

**REMEDIAL INVESTIGATION REPORT
WOOLFOLK CHEMICAL WORKS OPERABLE UNIT 5
FORT VALLEY, GEORGIA**

FINAL

**Prepared for:
U.S. Environmental Protection Agency
61 Forsyth Street SW, 11th Floor
Atlanta, Georgia 30303**

**Prepared by:
J.M. Waller Associates, Inc.
100 Hartsfield Centre Parkway, Suite 610
Atlanta, Georgia 30354**

**REMEDIAL ACTION CONTRACT (RAC) II LITE
IN EPA REGION 4**

**EPA CONTRACT NO. EP-S4-08-03
TASK ORDER 06**

AUGUST 2013



**J.M.WALLER[®]
ASSOCIATES, INC.**

A SERVICE DISABLED VETERAN OWNED SMALL BUSINESS

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Tim Bricker, P.E.
Project Manager

Date

Arnold Ostrofsky, P.E.
Program Manager

Date

DISTRIBUTION LIST

This document will be distributed to the following project participants:

Charles King
U.S. EPA Region 4
61 Forsyth Street S.W., 11th Floor
Atlanta, Georgia 30303

Michael Elster
Georgia Environmental Protection Division
2 MLK, Jr. Drive, S.E., Suite 1154
Atlanta, GA 30334

Arnold Ostrofsky
J. M. Waller Associates
100 Hartsfield Centre Parkway, Suite 610
Atlanta, GA 30354

Tim Bricker
Tetra Tech
2171 West Park Court, Suite E
Stone Mountain, GA 30087

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ACRONYMS AND ABBREVIATIONS

Σ3PC	Sum of p-26, p-50, and p-62 (three key Parlars)
°C	degrees Celsius
°F	degrees Fahrenheit
ABS	absorption fractions
ADAF	Age-dependent adjustment factor
ADD	average daily dose
AF	adherence factor
ALM	Adult Lead Model
ARAR	Applicable or Relevant, and Appropriate Requirements
AT	averaging time
atm-m ³ /mole	atmosphere per cubic meter per mole
ASTM	American Society for Testing and Materials
ATSDR	Agency for Toxic Substances and Disease Registry
ATV	alternative toxicity value
BAF	bioaccumulation factor
BCA	bias-corrected accelerated
BERA	baseline ecological risk assessment
BF	Brake fern
bgs	below ground surface
BHC	benzene hexachloride
BW	body weight
C	control
Ca	calcium
CaCO ₃	calcite
CalEPA	California Environmental Protection Agency
CCME	Canadian Council of Ministers of the Environment
Cd	cadmium
CDM	Camp, Dresser, and McKee
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act of 1980
CGC	Canadyne Georgia Corporation
CLP	Contract Laboratory Program
CLT	central limit theorem
cm ²	centimeters squared
CMC	Clay M. Corman
COPC	chemicals of potential concern

ACRONYMS AND ABBREVIATIONS (CONTINUED)

Cr	chromium
Cu	copper
CS	threshold abiotic media concentration
CSF	cancer slope factor
CSM	conceptual site model
DDD	dichlorodiphenyldichloroethane
DDE	dichlorodiphenyltrichloroethylene
DDT	dichlorodiphenyltrichloroethane
DI	deionized
DPT	direct-push technology
DQO	data quality objective
dw	dry weight
Eco-SSL	ecological soil screening level
ED	exposure duration
EF	exposure frequency
ERA	ecological risk assessment
ESAT	Environmental Services Assistance Team
ESL	ecological screening level
ET	exposure time
Fe	Iron
FI	fraction ingested
FOD	frequency of detection
FS	Feasibility Study
FVSU	Fort Valley State University
G	gram
GI	gastrointestinal
g/mole	gram per mole
HCH	hexachlorocyclohexane
HHRA	Human Health Risk Assessment
HI	hazard index
HQ	hazard quotient
IEUBK	Integrated Exposure Uptake Biokinetic
ILS	Integrated Laboratory Systems
IRS	incidental soil ingestion
J.M. Waller	J.M. Waller Associates, Inc.

ACRONYMS AND ABBREVIATIONS (CONTINUED)

K	potassium
Kd	soil-water partition coefficient
kg	kilogram
KM	Kaplan-Meier
Koc	organic carbon partition coefficient
Kow	octanol/water partition coefficient
L	liter
LOAEL	lowest-observed-adverse-effect level
LOEC	lowest-observed-effect concentration
MCL	maximum contaminant level
m ³ /kg	cubic meter per kilogram
mg	milligram
mg/cm ²	milligram per centimeter squared
mg/day	milligram per day
mg/kg	milligram per kilogram
mg/kg-day	milligram per kilogram per day
mg/L	milligram per liter
mg/m ³	milligram per cubic meter
mL/g	milliliter per gram
MVUE	minimum variance un-biased estimator
ND	non detect
No.	number
NOAEL	no-observed-adverse-effect-level
NOEC	no-observed-effect concentration
NRWQC	National Recommended Water Quality Criteria
OU	Operable Unit
p-26	Parlar 26
pP-50	Parlar 50
p-62	Parlar 62
PAH	polycyclic aromatic hydrocarbon
PCPI	Peach County Properties, Inc.
PCB	polychlorinated biphenyl
PEC	probable effect concentration
PEF	particulate emission factor
ppm	part per million

ACRONYMS AND ABBREVIATIONS (CONTINUED)

RA	remedial action
RD	remedial design
RfC	reference concentration
RfD	reference dose
RGO	remedial goal option
RI	Remedial Investigation
RME	reasonable maximum exposure
ROD	Record of Decision
RPM	Remedial Project Manager
RRS	risk reduction standard
RSL	Regional Screening Level
SA	surface area
SARA	Superfund Amendments and Reauthorization Act of 1986
SESD	Science and Ecosystem Support Division
SQL	sample quantitation limit
s.u.	standard unit
SVOC	semivolatile organic compound
TAL	Target Analyte List
TIC	tentatively identified compound
TCL	Target Compound List
TOC	total organic carbon
TRV	toxicity reference value
TSP	triple superphosphate
TT	technical toxaphene
UC	Upper Cretaceous
UCL	upper confidence limit
µg/dL	microgram per deciliter
µg/L	micrograms per liter
µg/m ³	micrograms per cubic meter
UGA	University of Georgia
URF	unit risk factor
USEPA	US Environmental Protection Agency
VOC	volatile organic compound
Woolfolk	Woolfolk Chemical Works
WT	weathered toxaphene

ACRONYMS AND ABBREVIATIONS (CONTINUED)

ww	wet weight
XRF	x-ray fluorescence

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EXECUTIVE SUMMARY

J.M. Waller Associates, Inc. (J. M. Waller) and its team member Tetra Tech, Inc. were tasked by United States Environmental Protection Agency (USEPA) Region 4 to conduct a fund-lead Remedial Investigation/Feasibility Study (RI/FS) for the Woolfolk Chemical Works (Woolfolk) Operable Unit (OU) 5. The purpose of this RI report is to characterize the nature and extent of contamination in sediments along the affected drainage pathway, delineate the source of arsenic detected in the sediments, identify potential contaminant migration pathways, prepare risk assessments (human health and ecological), and collect enough data to allow for the development of remedial action alternatives. A comprehensive FS identifying the cleanup options being considered for the OU5 site will be issued as a separate document.

This RI report was developed for sediment contamination and does not include assessment of other media, such as soil, surface water and groundwater. The interpretation of the data and information compiled for the OU5 site included in this RI report indicates that:

- Based on the data from investigations by USEPA and JM Waller of the OU5 site, the following conclusions can be made:
 - Concentrations of metals and pesticides in sediments exist at levels greater than RGOs. The individual contaminants are arsenic, lead, DDD, DDE, DDT and toxaphene.
 - The majority of contaminants exist adjacent to Ira Hicks Boulevard. Most of the individual contaminants above RGOs exist within the same footprint.
- Potential risks to human health from the OU5 site areas are in excess of USEPA acceptable limits or levels of concern based on the hazard indices and cancer and non-cancer risk thresholds. The cancer risks for the trespassers, groundskeeper, and utility worker receptors were within or below the USEPA risk range and all of the non-cancer hazard indices were less than the non-cancer benchmark of one (see Section 7.0).
- The results of the ecological risk characterization indicate that there may be unacceptable risks to ecological receptors from exposure to contaminated soils/sediment in OU5, but not necessarily from surface water or aquatic biota. RGOs were developed for individual contaminants.
- The phytoremediation treatability study showed that Chinese Brake ferns and Silver Ribbon ferns removed arsenic from Woolfolk site sediment under greenhouse conditions.

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1.0 INTRODUCTION

J.M. Waller Associates, Inc. (J.M. Waller) and its team member Tetra Tech, Inc. (Tetra Tech) were tasked by United States Environmental Protection Agency (USEPA) Region 4 to conduct a fund-lead remedial investigation/feasibility study (RI/FS) for the Woolfolk Chemical Works (Woolfolk) Operable Unit (OU) 5, located in Fort Valley, Peach County, Georgia. This RI report was prepared under the authority of the Comprehensive Environmental Response Compensation and Liability Act of 1980 (CERCLA) and the Superfund Amendments and Reauthorization Act of 1986 (SARA). The J.M. Waller Team prepared this document under Contract Number (No.) EP-S4-08-03, Task Order No. 06 in accordance with the USEPA Statement of Work (dated September 28, 2008) and the final RI/FS Work Plan (dated September 2012).

1.1 OBJECTIVES

The scope of this RI is to collect additional environmental information to characterize the nature and extent of sediment contamination along a drainage pathway defined as Woolfolk OU5, assess the potential risks to human health and the environment; and to collect sufficient information to allow for the development of remedial action alternatives for the FS. Therefore, the primary objectives identified for this RI are to:

- Characterize the nature and extent of contamination in sediments along this drainage pathway using relevant existing data and collection of additional data as needed.
- Identify and delineate the source of the arsenic detected in the sediments.
- Prepare risk assessments to determine whether arsenic or pesticide contaminants in sediments pose a current or future potential risk to human health and the environment.
- Identify the potential contaminant migration pathways.
- Collect sufficient data for development of a limited number of remedial action alternatives that are protective of human health and the environment and that satisfy applicable or relevant and appropriate requirements (ARARs).

The ultimate goal is to support the selection of a site remediation approach by developing the minimum amount of data necessary, and using this data in a well-supported Record of Decision (ROD).

1.2 REPORT ORGANIZATION

This RI report presents and evaluates relevant existing data from all data collected during the field investigation activities to characterize the nature and extent of arsenic or pesticide contamination.

Section 1 identifies the objectives of the Woolfolk OU5 RI report. Section 2 describes the Woolfolk history and previous investigation information. Section 3 discusses the site setting and physical characteristics. Section 4 describes the sampling and analysis activities and methodology. Section 5 describes the nature and extent of sediment contamination. Section 6 addresses the fate and transport of site-related contaminants. Section 7 presents the findings of the baseline human health risk assessment, and Section 8 presents the ecological risk assessment. Section 9 includes the phytoremediation pilot study. Section 10 presents the summary and conclusions. Section 11 includes the references.

2.0 SITE BACKGROUND

This section provides a description of the Woolfolk Superfund property including its location and boundaries, and a detailed history of Woolfolk activities.

2.1 SITE DESCRIPTION

The Woolfolk Superfund site is located at 126 East Main Street in Fort Valley, Peach County, Georgia. USEPA's ID number for this site is GAD003269578. The main site covered approximately 31 acres, including the former Woolfolk plant (approximately 18 acres), and the surrounding commercial areas where contamination has spread. Businesses formerly operating on the main property included SurePack, Inc., Georgia Ag Chem, Inc., and the Marion Allen Insurance and Realty Company.

The former Woolfolk facility is located adjacent to the central business district of Fort Valley near an area with mixed commercial and residential uses. According to the 2010 census, approximately 9,800 people live in the city limits. Residences are located west, south, and east of the former facility, with homes to the southeast adjoining a pecan orchard. Several businesses and light industrial sites are located along the north, northwest, and east ends of the former facility, including the Norfolk Southern Railroad tracks and station. The site is currently zoned for industrial use.

Over the lifetime of site operations, groundwater in the surficial Upper Cretaceous (UC) water table and UC confined aquifers beneath the site has been contaminated with pesticides and inorganics. Trace contaminants have been detected directly beneath the site in the deeper Tuscaloosa aquifer, and recent sampling has shown the presence of contaminants in monitoring wells located near the former facility. The city of Fort Valley public water wells, which draw water from the Tuscaloosa aquifer, have never shown contamination. Soil and buildings located at the former facility were contaminated with high levels of lead and arsenic. Residential parcels located near the facility also had high levels of arsenic in soil.

Contamination has also been found in a drainage pathway leading from the former facility to an upper tributary of Big Indian Creek. There are no wild and scenic rivers or designated wilderness areas in Peach County, and the site does not affect any coastal zones or coastal barriers. The site is located in a geographical area where endangered species including the Red-cockaded woodpecker (*Picoides borealis*), Kirtland's warbler (*Setophaga kirtlandii*), and Bald eagle (*Haliaeetus leucocephalus*) may reside. However, the site is not likely to affect endangered species because there are no records or reported sightings of these species within Peach County.

2.2 SITE HISTORY

The Woolfolk facility has been used for the production and packaging of organic and inorganic insecticides since 1910. In the 1940's, arsenic trichloride was reportedly produced at the facility for the War Production Board. Production was expanded during the 1950s to include the formulation of various organic pesticides, including dichlorodiphenyltrichloroethane (DDT), lindane, benzene hexachloride (BHC), toxaphene, and other chlorinated pesticides. These organic pesticides and other insecticides and herbicides were formulated, packaged, or warehoused at the facility.

In 1986-87, an interim soil remediation was completed by Canadyne Georgia Corporation (CGC) at the Woolfolk facility. Several buildings were demolished and approximately 3700 cubic yards of soil contaminated with lead and arsenic was excavated. All contaminated soil above a certain level of contamination was disposed of at a permitted hazardous waste landfill in Alabama. Other soils and debris were disposed underneath an on-site cap currently owned by CGC. CGC informed the Georgia Environmental Protection Division (EPD) of the investigations and cleanup activities.

Five OUs have been designated at the site. OU1 addresses contamination of groundwater emanating from the Woolfolk facility and extends beyond the Fort Valley city limits. The OU1 FS was completed in December 1993; a ROD for OU1 was issued in March 1994. The remedy for OU1 initiated groundwater delineation, collection of data on aquifer response for remediation, and the restoration of groundwater to prevent possible future exposure to contaminated groundwater. In May 1994, a Unilateral Administrative Order was issued to several potentially responsible parties to complete the remedial design (RD) or remedial action (RA). CGC has complied with the Order by implementing the RD activities. The most recent investigations have shown that although an on-site groundwater extraction and treatment system has been in operation for several years, the groundwater contamination has migrated off-site. At this time CGC has discontinued treatment activities. USEPA, under the Superfund program, has taken over this cleanup. The extent of the groundwater contamination and the effectiveness of this treatment system is currently being evaluated.

OU2 addresses soil contamination on properties located on Martin Luther King Drive and Oak Street, which were proposed for redevelopment. USEPA issued a ROD for OU2 in September 1995. In addition, deed restrictions were filed to prevent use of these properties for residential purposes. Redevelopment of these properties, completed in 1998, included a new library, the renovation of the Troutman House into a Welcome Center, and the renovation of an office building into the Adult Education Center.

OU3 addresses the former Woolfolk Chemical Works facility. Contamination was present in soils, the surface/stormwater sewer system and in buildings. The ROD for OU3 was signed in 1998. However, during the public comment period, CGC indicated that there were inaccuracies in the volume calculations regarding the amount of soil and debris to be excavated from the cap area. Because of sampling activities

conducted during RD, USEPA found that CGC was correct. The ROD for OU3 was amended in February 2004 with a recommendation to excavate contaminated soils in the source area. In 2009, an Explanation of Differences document was prepared to discuss increased amounts of contaminated soil and the area of the cover cap. Over 500,000 cubic yards of contaminated material was removed from OU3.

OU4 addresses contamination in attic dust and soils, and in sediments along the drainage ditch to Spillers Street. The ROD for OU4 was signed in August 2004. During the OU4 sampling activities, three attics and several residential properties were identified with arsenic concentrations exceeding emergency response thresholds. As a result, the properties were addressed by USEPA's emergency response branch.

OU5 addresses contamination along the drainage pathway originating at the end of Spiller Street extension, generally heading due south for 3 miles, and terminating south of Carver Drive Bridge into an upper tributary of Big Indian Creek. The drainage ditch near Spiller Street was originally part of OU4, but additional sampling and evaluation was needed before cleanup alternatives could be developed. OU5 was created to allow time for this additional work without delaying cleanup activity in OU4.

2.3 PREVIOUS INVESTIGATIONS OF OU5

In January 2005, USEPA Region 4 Science and Ecosystem Support Division (SESD) collected samples along Bay Creek and Big Indian Creek (and its tributaries) to evaluate for the presence of contaminants which may have originated from the Woolfolk site. The sampling included 9 sediment and 2 benthic macroinvertebrate samples. The sediment samples (WCW001 and WCW003 through WCW010) were analyzed for arsenic and lead. Arsenic and lead were detected in all of the sediment samples. The investigation was summarized in the *Field Sampling Investigation Report*, which is included in Appendix I (USEPA 2006).

In September 2007, USEPA Region 4 SEDS and USEPA Region 4 Environmental Services Assistance Team (ESAT) collected samples along Baptist Creek, Bay Creek and Big Indian Creek (and its tributaries) to further delineate contaminants from the Woolfolk site. The sampling included 9 sediment, 13 surface soil, 6 surface water and 3 fish sampling locations. The sediment samples (WC101 through WC125) were analyzed for target analyte list (TAL) metals, target compound list (TCL) organochlorine pesticides, grain size distribution, pH and total organic carbon (TOC). Two surface soil samples (WC-106 and WC-109) were analyzed for soil toxicity using the earthworm *Eisenia foetida*, and for sediment toxicity using the amphipod *Hyalella azteca*. The surface water samples (WC105, WC115, WC118, WC121, WC123 and WC125) were analyzed for TAL metals, TCL organochlorine pesticides, grain size distribution, pH, conductivity, temperature, turbidity, and dissolved oxygen (DO). Five surface water samples were analyzed for toxicity using the freshwater cladoceran, *Ceriodaphnia dubia*. The fish samples (WC-115, WC-118, WC-121 and WC-123) were analyzed for TAL metals, TCL organochlorine pesticides, percent

moisture, and total lipids. Arsenic, lead and DDT were detected in several samples. The investigation was summarized in the *Field Investigation Data Report (Step 6)* which is included in Appendix I (USEPA 2008a).

In September 2008, USEPA Region 4 SEDS collected samples along an unnamed tributary of Big Indian Creek in the vicinity of Ira Hicks Boulevard (formerly University Boulevard) and Carver Drive to evaluate for the presence of contaminants which may have originated from the Woolfolk site. The sampling included 23 sediment locations. The sediment samples (WC126 through WC155) were analyzed with x-ray fluorescence (XRF) offsite after drying in an oven. Arsenic was detected in 19 of the sediment samples. The investigation was summarized in the *Soil/Sediment Sampling for XRF Analysis* report which is included in Appendix I (USEPA 2008b).

In February 2009, USEPA Region 4 SEDS and Integrated Laboratory Systems, Inc. (ILS) collected samples along an unnamed tributary of Big Indian Creek in the vicinity of Ira Hicks Boulevard (formerly University Boulevard) and Carver Drive to evaluate for the presence of contaminants which may have originated from the Woolfolk site. The sampling included 5 sediment locations and 4 fern locations. The sediment samples (WC125, WC 127, WC 150A, WC155 and WC156) were analyzed in an offsite laboratory for arsenic by XRF, grain size distribution, pH and TOC. Arsenic was detected in all of the sediment samples. Phytoremediation pilot studies were conducted with the site sediment samples using three different species of ferns: Cretan brake (*Pteris cretica* 'Mayi'), ladder brake (*P. vittata*), and ebony spleenwort (*Asplenium platyneuron*) in the SEDS laboratory. Of the 15 plants tested, only one plant (a Cretan Brake fern) was effective at reducing arsenic concentrations in soils. The investigation was summarized in the *Arsenic Bioaccumulation in Ferns: A Pilot Study* report which is included in Appendix I (ILS 2010).

Several ecological risk assessment (ERA) and toxicity documents have been prepared for the area included in OU5. A detailed discussion of these documents is presented in Section 7 of this Final RI Report. The documents include the following:

- *Preliminary OU4 Ecological Risk Evaluation*, (CH2MHILL 1997);
- *Technical Memorandum, Step 3A of the Problem Formulation for the Baseline Ecological Risk Assessment (BERA)*, (CDM 2000);
- *Problem Formulation, Baseline Ecological Risk Assessment (Step 3)*, (ILS 2006);
- *Toxicity of Surface Water, Sediment, and Surface Soil Samples*, (USEPA 2007);
- *Study Design and Data Quality Objectives (DQO) Process (Step 4)*, (ILS 2007a);
- *Sampling and Analysis Plan (Step 4)*, (ILS 2007b);
- *Field Investigation and Data Report (Step 6)*, (USEPA 2008a);
- *Step 7 – Risk Characterization, Ecological Risk Assessment*, (ILS 2009).

- *Toxicity Assessment of Surface Soil and Sediment*, (USEPA 2010b).

The March 2009 Step 7 – ERA concluded that aquatic and terrestrial receptors were at risk from exposure from pesticides and metals at the Woolfolk site and recommended additional characterization of the sediments.

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3.0 SITE SETTING AND CHARACTERISTICS

The following sections present site-specific information on the physical and anthropological conditions which exist at and near the Woolfolk site.

3.1 GEOLOGY

The site geology presented here is summarized from the *Final Remedial Investigation Report, Woolfolk Chemical Works Site, Fort Valley, Georgia* (CH2M HILL, 1992). In general, the uppermost unit of the Fort Valley Plateau is clayey and sandy to pebbly undifferentiated residuum. The residuum is believed to be underlain by undifferentiated Paleocene-Middle Eocene sediments of the Mossy Creek Formation. These sediments are predominantly fine-to-medium grained sands with massive, white to grey, silty-sandy kaolin units. Previous investigations have referred to this unit as the Kaolin Unit, and it is present throughout the Woolfolk site. The off-site ditch and its banks are underlain by a soil layer described as Local Alluvial Land which consists of well drained sandy loam soils on drainage ways.

Unconsolidated Upper Cretaceous sediments underlie the undifferentiated Paleocene to Middle Eocene sediments at the Woolfolk site. This material is believed to be the Gaillard formation, which contains poorly sorted sands with flakes of muscovite and beds of maroon-stained clay (kaolin), and feathers out in the northwest portion of the Woolfolk site. The maroon staining may be associated with bioturbation of the sediments. However, differentiation between the three identified Upper Cretaceous units is based on large-scale depositional features such as cross bedding.

Below the Upper Cretaceous are Middle Cretaceous sands and clays that may be the Fort Valley area equivalent of Blufftown and Eutaw Formations. This formation, the Pio Nono Formation, is a part of the Oconee Group and consists of white, yellow, and maroon to light-green clayey sand to sandy clay. The lowermost geologic unit of interest at the Woolfolk site is the Tuscaloosa Equivalent. The term Tuscaloosa Equivalent has been adopted because this unit is a facies equivalent of the Alabama-West Georgia Tuscaloosa Formation in the Fort Valley area.

3.2 HYDROGEOLOGY

The investigation of groundwater contamination is addressed in OU1. The discussion that follows is included for information purposes only.

The groundwater beneath the site has been divided into four layers. The first (the surficial aquifer), second (UC water table), and third (UC confined aquifers) layers are not in use. The fourth layer, the Tuscaloosa aquifer, is the source for the city of Fort Valley's water supply. The closest city well is located

within 500 feet of the site, north of the railroad on the edge of the groundwater plume. Monitoring of the city wells since 1985 has shown no contaminant concentrations above maximum contaminant levels (MCLs).

The surficial aquifer is groundwater in sands located approximately 35 feet below land surface. The groundwater in the surficial aquifer is classified as Class IIb and is considered a potential drinking water source. However, groundwater from the surficial aquifer is not currently being used as a potable water source. The groundwater flow in the surficial aquifer is towards the southeast. Approximately 75 feet below land surface and below the surficial aquifer is the surficial perching unit. This is a clay unit, consisting primarily of kaolin, and is continuous below the former facility except to the north.

The UC consists of the UC water table and the UC confined aquifer. The UC water table is located 105 to 125 feet below land surface, below the surficial perching unit. The UC water table is composed of clayey and silty sands. The groundwater in the UC water table flows north and northeast below most of the former facility, and to the east in the southeast portion of the facility. The UC confined aquifer, located in a layer 125 feet below land surface, is a 20-foot-thick layer composed of sand. The groundwater found in the UC confined aquifer is also classified as Class Iib, similar to the surficial aquifer, but is not currently a source of drinking water.

The Tuscaloosa aquifer is approximately 20 feet thick and is composed of alternating layers of sand, clay, and clayey sand beneath the UC confined aquifer. USEPA classifies the groundwater from the Tuscaloosa aquifer as Class IIa, which is considered a current drinking water source. Hydraulic head measurements in the Tuscaloosa aquifer beneath most of the former facility indicate groundwater flow toward the southeast.

These measurements also show a reversal in groundwater flow direction in and beyond the northwestern part of the former facility. The direction of flow toward the northwest is consistent with the expected effects of pumping groundwater from the Tuscaloosa aquifer by the city of Fort Valley's wells.

Hydraulic and lithological data indicate the potential for groundwater flow vertically downward beneath the former facility. The general nature of the perched surficial unit observed through the hydraulic and lithological data makes it likely for groundwater to leak downward through this unit. Downward flow is also indicated by a comparison of the hydraulic-head gradients of the UC aquifers and the Tuscaloosa aquifer. Fort Valley currently operates several municipal wells that supply drinking water to local residents. These wells are tapped into the Tuscaloosa aquifer that also underlies the site. Future use of groundwater resources by the city of Fort Valley is expected. In addition, the Georgia EPD considers each of the aquifers underlying the site to be a valuable resource.

3.3 DRAINAGE – SURFACE WATER

Regionally, the Woolfolk site is located in the western edge of the Altamaha Basin, one of the major water basins of Georgia. In addition, Woolfolk site is just south of the fall line between the Piedmont and Coastal Plain Physiographic districts; however, Woolfolk also exists in an anomalous area characterized as the Fort Valley Plateau. In general, the Fort Valley Plateau is characterized by fewer streams and less local relief. Elevations range from 530 feet msl at the former Woolfolk Chemical plant to 370 feet msl at the Carver Drive bridge. Examination of the OU5 topographic features did not show any significant perennial surface water; however, an ephemeral water feature exists on site. This ephemeral water feature, which is assumed to be the main contaminant pathway, originates at the southern end of Spillers Street extension and continues south under a railroad bridge, through a culvert at Ira Hicks Boulevard, and under another bridge at Carver Drive before flowing into an upper tributary of Big Indian Creek. The total length of the ephemeral water feature is approximately 3 miles long.

3.4 CLIMATE

This section is based on data collected from 1961 to 1990 by the Southeast Regional Climate Center at the Macon Airport, Georgia Station, located approximately 20 miles northeast of the site. Regional average annual precipitation is 44.65 inches, with the highest monthly precipitation occurring in March (4.79 inches) and February (4.74 inches), and the lowest average precipitation occurring in October (2.18 inches). The area receives an average of 1.4 inches of snow annually. Average temperatures range from 45.5 degrees Fahrenheit (°F) in January to 81.5 °F in July.

3.5 DEMOGRAPHICS AND LAND USE

The J.W. Woolfolk Company owned and operated the Woolfolk facility from 1926 until 1941, when it dissolved and conveyed its assets to Woolfolk Chemical Works, Ltd. Woolfolk Chemical Works, Ltd. reorganized into the corporation Woolfolk Chemical Works, Inc. in 1972. In 1977, Reichold, Limited acquired all of Woolfolk's stock, pursuant to a stock purchase agreement. The stock purchase agreement was assigned to Canadyne Corporation, a wholly-owned subsidiary of Reichold, Limited. In 1984, Woolfolk Chemical Works changed its name to CGC. Also in 1984, most of the facility was sold to Peach County Properties, Inc. (PCPI). PCPI owns most of the former Woolfolk property and leased most of the property to its affiliate, SurePack, Inc. (later SureCo, Inc.), which formulated and packaged pesticides at the facility from 1984 until 1999. Another portion of the property was leased to Georgia Ag Chem, which operated a warehouse and distribution facility. Marion Allen Insurance and Realty Company also own one parcel located northwest of the facility.

Current land use near the former facility is industrial, commercial, and residential. The majority of the former facility consists of buildings, storage tanks, and stockpiles of equipment.

OU2 consisted of residential properties prior to its designation as part of the Woolfolk Superfund site. Since contamination at OU2 has been cleaned up, the former residential properties have been put into reuse as the Peach County Public Library, and the Fort Valley Welcome Center (the Troutman House). OU4 properties were previously used for residential and commercial use, and these uses continued after remediation activities were completed.

With the exception of the areas requiring restricted use (the former capped area, CI and C2), a similar mix of future land use is anticipated for the OU3 properties, as residents and businesses will continue to occupy the adjacent properties after remediation activities are completed. In addition, based on discussions with city officials, anticipated future land use for OU3 may include commercial and/or recreational uses. Citizens have also expressed interest in developing residential areas to the west and south of OU3. These residential redevelopment plans will be coordinated with OU3 cleanup plans to ensure the protection of human health and the environment.

The Woolfolk Superfund site was a Superfund Redevelopment Initiative Pilot Site in 2002. The city of Fort Valley was awarded \$50,000, and used this resource to study and report potential site reuse possibilities to USEPA. USEPA worked with the community to plan the redevelopment of the library, welcome center, and adult education center at OU2. The Superfund Redevelopment Initiative and USEPA Region 4 provided the city of Fort Valley with resources to host a community-based process to develop future land use plans that would encourage long-term stewardship at the site. A reuse planning report was completed in 2007. The report focused on the reuse of OU3 once the cleanup is completed. The city of Fort Valley has also received a Technical Assistance Grant to assist the community with technical issues related to the site.

The USEPA has issued a Brownfields Grant to the city of Fort Valley for redeveloping the site. Under a separate Superfund redevelopment grant issued by USEPA, the city of Fort Valley has also approached Georgia Institute of Technology to evaluate both current and future land use scenarios, and to provide a design that integrates future land use with redevelopment under the Brownfields initiative.

4.0 FIELD INVESTIGATION

Field events in support of the OU5 RI occurred in several different mobilizations from 2005-2011. Discussions of the 2005-2008 events are described in previous reports; this section will focus on the 2009 and 2011 events. Since OU5 is considered a surface water drainage pathway, the investigation focused on sediment and receptors potentially exposed to sediment. All samples collected during this investigation were defined as sediment. This also included subsurface samples. Sampling locations are shown on Figures 4-1 through 4-4.

4.1 METHODOLOGY

The existing USEPA site trailer located on the original Woolfolk property was used as the J.M. Waller team operations office. USEPA emergency response contractor, Clay M. Corman (CMC), provided a person to assist the J.M. Waller team in operation of the XRF field instrument for measurement of arsenic in collected samples. The XRF instrument was a Thermo Scientific NITON XL3p Analyzer, calibrated and maintained by CMC personnel.

Stainless steel hand augers were used to collect sediment samples for the August 2009 through December 2009 field events, while direct push technology (DPT) was used for the collection of sediment samples for the April 2011 field event. Because of the distance between sample locations and the J.M. Waller team operations office, decontamination was performed between depths at each location using five gallon plastic buckets with appropriate decontamination reagents as described in the USEPA-approved J.M. Waller Sampling and Analysis Plan and Quality Assurance Project Plan.

4.2 SURFACE AND SUBSURFACE SEDIMENT SAMPLING

The initial sample locations were selected based on previous data collected by USEPA personnel in January 2005 and October 2007 (USEPA, 2006 and USEPA, 2007). Figure 4-1 shows an overview of the sampling locations. Figures 4-2 through 4-4 show sampling locations in three distinct areas:

- Area 1 is the area adjacent to Ira Hicks Boulevard (formerly University Boulevard), shown on Figure 4-2,
- Area 2 is the area east of Carver Drive, shown on Figure 4-3, and
- Area 3 is the area adjacent to the Carver Drive Bridge, shown on Figure 4-4

4.2.1 Round 1 Sampling

The first round of sediment sampling was conducted in August 2009 (Round 1). The field screening level was set as the previously established site screening level for arsenic of 33 milligrams per kilogram (mg/kg) (USEPA, 2006). Sampling locations were laid out along transects oriented perpendicularly across the surface water flow pathway. Undisturbed sediment at the center of each transect was measured with the XRF instrument. If arsenic concentrations exceeded 33 mg/kg, additional sampling locations were selected laterally away from the surface water pathway at 10, 25 and 50 foot intervals on either side of the original sample location. This procedure was performed approximately every 200 feet throughout the drainage area. At some locations, additional transects were established upstream or downstream of the original transects to horizontally delineate the sediment.

Surface sediment samples were collected from the surface, and subsurface sediment samples were collected from 0 - 0.5 foot, 1.5 - 2 feet, and 2.5 - 3 feet. At three locations samples were also collected at 3.5 - 4 feet and 4.5 - 5 feet to delineate the XRF data. Each sediment sample was prepared according to USEPA Method 6200 by placing the sample into a plastic bag, mixing thoroughly and measuring with the XRF instrument for arsenic concentration. A total of 166 samples were collected from the following intervals: 45 surface sediments; 46 samples from the 0- to 0.5-foot interval, 46 samples from the 1.5- to 2-foot interval, 23 from the 2.5- to 3-foot interval, 3 from 3.5- to 4-foot interval, and 3 from 4.5- to 5-foot interval.

Following the XRF field screening, 62 samples were placed in laboratory-supplied bottles, preserved as specified in USEPA Document SW-846, transported to Contract Laboratory Program (CLP) laboratories, and analyzed for TCL organochlorine pesticides and TAL metals. Soil pH measurements were performed using USEPA Method 9045D on the same subset of samples that were sent to the fixed-base laboratory.

4.2.2 Round 2 Sampling

After the data from Round 1 was evaluated, USEPA requested that another round of sampling be performed. The second round of sediment sampling was conducted in late November and early December 2009 (Round 2). The goal of this additional investigation was to evaluate data gaps found during the Round 1 sampling event, especially the area north of Ira Hicks Boulevard. A total of 186 samples were collected during Round 2, from the following intervals: 59 surface sediments; 59 samples from the 0- to 0.5-foot interval, 48 samples from the 1.5- to 2-foot interval, 12 from the 2.5- to 3-foot interval, 3 from 3.5- to 4-foot interval, and 5 from 4.5- to 5-foot interval.

Following the XRF field screening, 12 samples were placed in laboratory-supplied bottles, preserved as specified in USEPA Document SW-846, transported to CLP laboratories, and analyzed for TCL

organochlorine pesticides and TAL metals. Soil pH measurements were performed using USEPA Method 9045D on the same subset of samples that were sent to the fixed-base laboratory.

4.2.3 Round 3 Sampling

Another round of sampling was conducted in April 2011 (Round 3). During Round 3, the goal was to further vertically delineate the transects north of Ira Hicks Boulevard, based on results from the Round 1 and Round 2 sampling events. A DPT Rig was used to collect soil samples at locations that were established along the same areas previously investigated. The soil borings were advanced up to 10 feet below land surface or until water was reached or XRF screened soil sample results were non-detect.

During Round 3, 129 samples were collected from the following intervals: 1 sample from the 0- to 1-foot interval, 14 samples from the 1- to 2-foot interval, 27 from the 2- to 3-foot interval, 26 from 3- to 4-foot interval, 22 from 4- to 5-foot interval, 5 from the 5- to 6-foot interval, 8 from 6- to 7-foot interval, 9 from 7- to 8-foot interval, 9 from 8- to 9-foot interval, and 8 from 9- to 10-foot interval.

Following the XRF field screening, 23 samples were placed in laboratory-supplied bottles, preserved as specified in USEPA Document SW-846, transported to CLP laboratories, and analyzed for TCL organochlorine pesticides and TAL metals. Soil pH measurements were performed using USEPA Method 9045D on the same subset of samples that were sent to the fixed-base laboratory.

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5.0 NATURE AND EXTENT OF CONTAMINATION

The following section discusses the nature and extent of contamination for OU5 based on the results of chemical analyses performed on sediment samples collected during the 2005-2011 field investigation events. Historical site sampling data were also used where applicable.

Field XRF screening results are shown in Tables 5-1 and 5-2 and Figures 5-1 through 5-3. Surface and subsurface sediment analytical results are shown in Tables 5-3 and 5-4 and Figures 5-4 through 5-29. Frequency of detection information is shown in Table 5-5. Analytical laboratory reports are included in Appendix II.

5.1 SUSPECTED SOURCE AREAS

Previous investigations identified the potential source of metals and pesticides in OU5 sediments as being from routine operations on the Woolfolk site. Another possible source could have been from applications of pesticides on adjacent agricultural property. Due to the lack of definitive evidence of an additional source area(s), and considering the distribution of sediment contaminants in relation to horizontal and vertical groundwater gradients in the subsurface, the metal and pesticide sediment contamination delineated by the RI between the former Woolfolk property and tributaries of Big Indian Creek was interpreted to be a plume attributable to the Woolfolk site. All downgradient metal and pesticide sediment contamination identified by this RI is located within a narrow band that generally follows the ephemeral drainage pathway from the former Woolfolk property to tributaries of Big Indian Creek. It is unknown if multiple releases over time may have occurred at Woolfolk, if contaminants originated from application of pesticides on adjacent agricultural property or if natural attenuation processes may be degrading contaminants to varying degrees throughout the plume.

5.2 SEDIMENT

5.2.1 Nature of Sediment Contamination

Analytical results from RI sediment sampling detected concentrations of arsenic, lead, DDD, DDE, DDT and toxaphene above remedial goal options (RGOs). A total of 167 sediment locations were evaluated with a total of 504 samples collected from varying depths. All 504 samples were screened in the field for arsenic with an XRF field instrument. Of the 504 samples, 136 samples were sent to CLP laboratories for TAL metals and TOC analysis (Method ILM05.4) and 126 were sent to CLP laboratories for TCL organochlorine pesticides analysis (Method SOM01.2). Of the 504 samples, 94 samples were greater than the 200 mg/kg RGO for arsenic, 20 samples were greater than the 260 mg/kg RGO for lead, 3 samples were greater than the 1.03 mg/kg RGO for dichlorodiphenyldichloroethane (DDD), 5 samples were greater than the 1.03 mg/kg RGO for dichlorodiphenyldichloroethylene (DDE), 19 samples were

greater than the 1.03 mg/kg RGO for DDT, and 15 samples were greater than the 6.7 mg/kg RGO for toxaphene.

5.2.2 Horizontal Extent of Sediment Contamination

The initial RI sediment sampling efforts focused on identifying the horizontal extent of metal and pesticide contamination in the sediments, which consists of concentrations of arsenic, lead, DDD, DDE, DDT and toxaphene in excess of RGOs. The majority of contamination exists adjacent to Ira Hicks Boulevard in a plume approximately 500 feet wide and 4,000 feet long adjacent to the ephemeral stream. Contamination also exists at two additional locations, east of Carver Drive and adjacent to the Carver Drive Bridge, but the occurrence and concentrations are much less than the Ira Hicks Boulevard area. The streambed north of Ira Hicks Boulevard is relatively flatter and wider than the rest of the study area. It appears that contamination is higher in this area as soil particles settled when the surface water slowed down and spread out.

Sediment field XRF and analytical data collected during the RI was input into CTech Development Corporation's Mining Visualization System (MVS) software for three-dimensional modeling and visualization of metals and pesticides. MVS was used to create color contour maps of the data to aid with data interpretation and presentation. Figures 5-1 through 5-21 show the extent and distribution of metals (arsenic and lead) and pesticides (DDD, DDE, DDT and toxaphene), respectively. Each figure represents the maximum value for each constituent detected at each location.

5.2.3 Vertical Extent of Sediment Contamination

The final RI sediment sampling effort focused on identifying the vertical extent of metal contamination in the sediments, which consists of concentrations of arsenic, lead, DDD, DDE, DDT and toxaphene in excess of RGOs. A total of 31 locations were investigated along 8 transects perpendicular to the ephemeral stream north of Ira Hicks Boulevard. The plume extends up to 10 feet deep in the middle of the streambed.

The vertical extent and distribution of XRF arsenic contamination is shown on eight cross sections. The locations of the cross sections are presented on Figure 5-22 and the individual cross sections, A-A' through G-G', are presented on Figures 5-23 through 5-29, respectively.

6.0 CONTAMINANT FATE AND TRANSPORT

Fate and transport within the OU5 study area focuses on the sediment contamination – its sources, migration pathways, and behavior during transport to potential human and ecological receptors. There are several mechanisms by which contaminants may migrate at the OU5 study area. Migration into the air can occur via volatilization or dust generation. Migration into groundwater can occur by percolation of rainwater into the sediment. Transport to the stream in the OU5 study area can occur via surface water runoff. The mechanisms of migration for the contaminants of concern detected at the OU5 study area are discussed in detail below. Estimates of the physical and chemical properties of the contaminants of concern which may affect contaminant migration are presented in Table 6-1.

6.1 CONTAMINANT MIGRATION

6.1.1 Air Migration

Generally, volatilization from soil and/or water into air may be an important transport mechanism for the organic chemicals with Henry's Law Constants greater than 10^{-5} atmosphere per cubic meter per mole ($\text{atm}\cdot\text{m}^3/\text{mole}$) and molecular weights less than 200 grams per mole (g/mole). Henry's Law Constant is defined as the ratio of the partial pressure of a compound in air to the concentration of the compound in water at a given temperature under equilibrium conditions. None of the four organic contaminants of concern (DDD, DDE, DDT and toxaphene) meet these two criteria. Therefore, volatilization of these organic compounds is not considered an important release mechanism at this site. The air release mechanism for pesticides and inorganics at the OU5 site would be associated with the movement of particulate matter as airborne particles generated by wind blow dusts from contaminated surface soils.

Fugitive dust emissions from wind or mechanical disturbances may occur from unpaved or unvegetated areas of the site. The environmental factors that influence wind erosion are wind speed, moisture content, vegetative cover, and soil composition. Because the environmental factors at the OU5 site are at times and places conducive to wind erosion, each of the contaminants of concern detected in surface soil is susceptible to migration via fugitive dust generation. Release of arsenic into the atmosphere from anthropogenic sources can result from agricultural chemicals such as those produced at the Woolfolk site. Most human releases are to land or soil, primarily in the form of pesticides or solid wastes. The pesticides detected in sediment at the OU5 site are weakly volatile due primarily to their relatively high molecular weights and low water solubilities, which tends to make them adsorb to soils and sediments.

6.1.2 Sediment Migration

Contaminants present in sediment soils may leach to the underlying aquifer. Many factors influence the rate of contaminant movement through soils. These include the physical/chemical properties of the contaminants (e.g., solubility, density, viscosity, partition coefficients), and the physical/chemical properties of the environment (e.g. rainfall percolation rate, soil permeability, porosity, particle size distribution, organic carbon content). Because all these factors can affect the rate of contaminant movement through soils, it is very difficult to predict such movement. However, based on the data collected in this investigation some gross generalizations of this movement can be made.

Sorption of a chemical to soil particles is the only significant hindrance of contaminant migration in sediment at the OU5 site. If it were not for sorption, rainfall recharge, and soil permeability at this site, the contaminants of concern would readily move downward through the sediment. Sorption of contaminants is generally described by their soil-water partition coefficients (K_d). The K_d is the mass of contaminant on the solid phase per mass of solid phase divided by concentration of solute in solution. The K_d for metals is highly dependent on pH. Under low pH conditions, the K_d for metals is significantly reduced thus rendering them more mobile in soils.

Estimates of the K_d for the contaminants of concern are in Table 6-1. The total organic carbon results and fraction of organic carbon calculations are in Table 6-2. K_d values greater than 10 milliliter per gram (mL/g) indicate significant sorption potential and thus limited propensity to leach from sediment to groundwater. All four organic contaminants of concern (toxaphene, p,p'-DDD, p,p'-DDE, and p,p'-DDT) and two inorganic contaminants of concern (lead and arsenic) at this site have K_d values greater than 10 mL/g. Therefore, these contaminants are expected to readily sorb to sediment particles at the OU5 site. Leaching of these contaminants of concern from the sediment to groundwater at OU5 is expected to be relatively limited due to their high propensity for sorption.

6.1.3 Surface Water Migration

Contaminant migration into surface waters at the OU5 site may occur through surface runoff and/or through groundwater discharge. Upon reaching these surface water pathways, the contaminants may remain in the water column, volatilize, or sorb to bottom or suspended sediments. Volatile organic compounds (VOCs), none of which were identified as contaminants of concern, tend to quickly volatilize into the atmosphere upon reaching surface water and for this reason are rarely observed at detectable concentrations in water samples. All of the semivolatile organic compound (SVOC) contaminants of concern, including the pesticides, have low water solubilities, high organic carbon partition coefficient (K_{oc}) values, and tend to associate with sediments.

The potential behavior of the inorganic contaminants of concern in the surface waters at the OU5 site is affected by pH, temperature, and hardness. Inorganic compounds can occur in aquatic systems as dissolved ions, dissolved complexes with organic and inorganic chemicals, colloids, or particulates. The solubility and mobility of the inorganic contaminants is enhanced by their ability to form complexes with humic and fulvic acids, carbonates, hydroxides, and phosphates. Because of the high K_d coefficients, the majority of the inorganic contamination would be expected to sorb to sediment particles when released to the drainage system; however, the geochemical processes in surface water identified above will also likely allow for some of the inorganic contamination to remain or dissolve in the water column. How much dissolves and how much sorbs to sediment particles will depend on the geochemical conditions of the drainage ditch area, which may vary temporally and spatially. Again, however, the sorbing of contaminants to sediment particles does not preclude the migration of these contaminants since the contaminated sediments may still be conveyed downstream.

6.2 CONTAMINANT PERSISTENCE

Persistence is the measure of how long a chemical will exist in the environment before it degrades or transforms, either chemically or biologically, into some other chemical. Some of the factors which affect the persistence of a chemical include the state of the chemical, the availability of the chemical, exposure to sunlight, oxygen availability, the types and quantities of microorganisms present, availability of nutrients, temperature, pH, as well as the presence of other chemicals which may inhibit or enhance degradation. Usually, persistence is expressed in terms of a chemical half-life and can be on the order of days, weeks, or years. Because of the many complex factors which may affect persistence, the actual rate of chemical degradation is very difficult to predict for a given chemical at a given site, especially without the benefits of any degradation data collected from site-specific field studies.

Of all the contaminants of concern at the OU5 site, the inorganics have the least potential to degrade in all media; therefore, will likely persist the longest at the site. In fact, these contaminants, under ordinary conditions, will likely persist indefinitely (for all practicable purposes). Arsenic in the sediment exists in various oxidation states and chemical species, depending upon soil pH and redox potential. Other chemicals which also have low degradation potential include the pesticides and PCBs.

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7.0 HUMAN HEALTH RISK ASSESSMENT

The Human Health Risk Assessment (HHRA) was developed in accordance with USEPA guidance to determine whether detected concentrations of chemicals within the study area pose a significant threat to potential human receptors under current and/or future land use. The potential risks to human receptors are estimated based on the assumption that no actions are taken to control contaminant releases.

7.1 HAZARD IDENTIFICATION

The objective of the hazard identification is to present the data available to assess site risks, outline the approach used to summarize site data, and identify contaminants of potential concern (COPCs). The following sections describe the methods that were used for data reduction, data evaluation, exposure units (EUs), and selection of COPCs:

7.1.1 Available Data

Table 7-1 presents a list of sampling locations used in the HHRA. An original investigation for the Woolfolk site was conducted in September 2007. Abiotic samples collected during the site investigation included 14 surface soils (i.e., 11 soil samples, 1 duplicate, and 2 background samples), 5 subsurface soils, 10 sediments (i.e., 7 site samples, 1 duplicate, and 2 background samples), and 7 surface water samples (i.e., 4 site waters, 1 duplicate, and 2 background samples). At the request of the USEPA Remedial Project Manager (RPM), 25 additional surface soil and sediment samples were collected by USEPA Region 4 personnel in September 2008 (USEPA, 2008b) and analyzed for arsenic via an XRF.

In August, November, and December 2009, XRF was used to analyze 104 sediment samples (plus 1 duplicate) and 105 sediment samples that were 0-0.5 foot below ground surface (bgs) (plus 6 duplicates) for arsenic, in addition to numerous subsurface samples. Fixed-base laboratory analyses were also performed on 36 co-located 0-0.5 foot bgs samples. As part of the invertebrate soil/sediment toxicity tests conducted, samples were also collected in April 2011. Two background sediment (0-0.5 foot bgs) samples were collected, as well as four site sediment and two site soil samples (0-0.5 foot bgs) plus one duplicate. In April 2011, XRF was used to analyze 31 site soil samples at various depth intervals ranging from 0-10 feet bgs. Additionally, 21 site subsurface soil samples (plus 2 duplicates) were also taken for fixed-base laboratory analysis during this sampling event.

The focus of this HHRA is on soil and sediment exposure only. There are no plans to sample surface water in the upper tributary of Big Indian Creek as part of this RI; therefore, surface water is eliminated from further consideration. Although the site drainage ditch is sometimes covered with water and could be considered sediment at those times, the exposure pathways for both media in the HHRA are the same. Therefore, the surface soil (0-0.5 foot bgs) and sediment data are evaluated together as surface soil. Soil

samples below 0.5 foot bgs were considered subsurface soil and were combined with the surface soil and sediment samples for the evaluation of total soil.

Consistent with the approach used in the Woolfolk Chemical Ecological Risk Assessment (J.M. Waller, 2010), the 2009 XRF data showed good correlation with the samples analyzed at the fixed-base laboratory. As such, for samples that had only XRF data, these results were used (i.e., when a sample was analyzed by both XRF and the fixed-base laboratory, only the fixed-base laboratory data were used). Based on the correlation between 2009 XRF and fixed-base laboratory data, the same was assumed for the 2011 XRF samples. Therefore, the 2009 approach for inclusion of XRF samples was also used for the 2011 XRF samples. XRF samples had no detection limits associated with them; therefore, if arsenic was not detected in an XRF sample, the sample was not used.

7.1.2 Exposure Units

OU5 covers over 5 miles and distances between different groups of sampling locations is far greater than a potentially exposed individual is likely to travel. Current land uses were based on the Peach County zoning classifications, shown on Figure 7-1. Therefore, the site was divided into subareas to allow for a more meaningful evaluation of contamination and potential exposure. The approach to developing EUs for evaluation in the HHRA was to identify as large an area as reasonable to maximize the number of samples per EU. A total of four EUs were designated based on the proximity of sample locations and reasonably anticipated land uses along the drainage ditch. Therefore, each EU represents a reasonable exposure area to potential receptors. Figures 7-2 through 7-5 present the locations of EUs and their associated sampling locations.

7.1.3 Data Evaluation

This section presents the approach that was followed to prepare the analytical data for use in the COPC screening process and for the calculation of risks.

7.1.3.1 Data Reduction

Data reduction involves the evaluation of data qualifiers and their potential use in the HHRA process and describes the treatment of duplicate and co-located samples. The following guidelines were used in developing the data sets to evaluate risk associated with OU5:

- If an analyte was not detected in any sample from a given medium, it was not considered further for that medium.
- Analytical results with an “R” qualifier (indicating that the data was rejected during the validation process) were not retained in the data set.

- All “U” qualified data represent samples for which the analyte was not present or was below the sample quantitation limit (SQL). These data are considered non-detects (NDs) and were retained in the data set at the full detection limit.
- Analytical results with a “J” qualifier indicates that the reported value is estimated because the analyte is detected at a concentration below the SQL or for other reasons. These data are considered positive detections and were retained in the data set at the measured concentration.

When field duplicate samples were collected, the following approach was used to calculate the concentrations to be evaluated in the HHRA:

- If the analyte was detected in both the original (primary) sample and the field duplicate, the average of the two detected concentration was used.
- If the analyte was detected in either the primary or duplicate sample and was ND in the other sample, the detected concentration was used.

If the analyte was ND in the primary and duplicate sample, the average of the two SQLs was used.

7.1.3.2 Evaluation of Weathered Toxaphene

Weathered toxaphene (WT) is discussed separately here since it is a special case and its evaluation requires additive discussion. Technical toxaphene (TT), which refers to toxaphene as manufactured, is a mixture of up to 800 different congeners and is persistent in the environment. Once in the environment, TT can undergo weathering due to biotic and abiotic processes. The congeners found in TT vary in number, identity, and toxicity compared to those found in toxaphene in the environment or WT. Several of the TT and WT congeners are also known as Parlars and have been given a Parlar number for easier identification. Toxaphene weathers differently in different environmental media to yield varying congener mixtures (Simon and Manning, 2006). Of the many WT congeners, only a limited number have been found to be persistent in the human body. Of those persistent congeners found in the human body from WT, 95 percent are attributable to Parlar 26 (p-26), Parlar 50 (p-50), and Parlar 62 (p-62) (Simon and Manning, 2006).

Technical toxaphene and several toxaphene Parlars were detected in OU5 surface soil. In order to screen for and evaluate the toxicity of the three primary congeners of WT, a non-cancer reference dose has been developed (Simon and Manning, 2006) and used as a screening toxicity value for the summation of p-26, p-50, and p-62. For each EU, the p-26, p-50, and p-62 concentrations were summed on an individual sample basis to arrive at a sum of the three key Parlars, herein referred to in this HHRA as $\Sigma 3PC$. As noted previously, non-detects were also included in the $\Sigma 3PC$. It should also be noted that TT is also evaluated in this HHRA.

7.1.4 Contaminant of Potential Concern Screening

Based on the history of releases from the facility, contamination is present in environmental media at OU5. The concentrations of the observed contaminants were screened against risk-based criteria and background levels (for metals) to determine if these contaminants warrant further evaluation in the HHRA.

Tables 7-2 through 7-9 present the data summaries for EUs 1 through 4 for the surface soil and total soil, respectively. The summary tables present the following information:

- List of detected analytes.
- The range of detected concentrations.
- The location of the maximum detected concentration.
- The frequency of detection (FOD).
- The range of SQLs.
- The screening toxicity value – USEPA Regional Screening Level (RSL) (USEPA, 2012b).
- Whether the analyte was selected as a COPC.

The COPC screening process was conducted in accordance with USEPA Region 4 guidance (USEPA, 2000a). The maximum detected concentrations in soil were compared to the USEPA RSLs (USEPA, 2012b). The cancer based RSLs were set at a target cancer risk of 1E-06. The noncancer based RSLs were set at a target hazard quotient (HQ) of 0.1. The residential soil RSLs were used for selecting COPCs in soil. The residential soil RSLs are based on exposure assumptions indicative of exposure associated with residential backyards and are considered conservative for this HHRA since residential exposure is not expected to occur in the future.

If the maximum detected concentration was less than the RSL, the analyte was eliminated from further consideration in the HHRA. If the maximum concentration exceeded the RSL, the contaminant was identified as a COPC. Further, any member of a contaminant class (e.g., polycyclic aromatic hydrocarbons [PAHs]) that has other members identified as COPCs was also retained as a COPC (USEPA, 2000a).

For metals in surface soil exceeding the RSLs, a comparison to site-specific background levels was performed. The background data used in the comparison was from the 0 to 0.5 foot bgs depth range and was collected from September 2007 and April 2010. The site surface soil maximum concentrations were compared to two times the average background concentration (USEPA, 2000a). If the site surface soil maximum was less than two times the average background level, the metal was eliminated from consideration as a COPC. Due to a lack of subsurface data from background locations, those metals eliminated as COPCs for surface soil were also eliminated as COPCs for total soil.

The analytes detected in the soil included pesticides, PAHs, VOCs, SVOCs, and inorganics. The following table presents those detected contaminants in each EU that exceeded the residential soil RSLs:

EU1		EU2	EU3	EU4
4,4'-DDD	Benzo(a)anthracene	Dieldrin	Aluminum	Benzo(a)anthracene
4,4'-DDE	Benzo(a)pyrene	Aluminum	Arsenic	Benzo(a)pyrene
4,4'-DDT	Benzo(b)fluoranthene	Arsenic	Chromium	Benzo(b)fluoranthene
Aldrin (total soil only)	Benzo(k)fluoranthene	Chromium	Cobalt	Benzo(k)fluoranthene
alpha-BHC	Chrysene	Iron	Iron	Chrysene
beta-BHC	Dibenz(a,h)anthracene	Vanadium	Manganese	Indeno(1,2,3-cd)pyrene
delta-BHC	Indeno(1,2,3-cd)pyrene		Vanadium	Aluminum
gamma-BHC	Aluminum			
alpha-Chlordane	Antimony			
gamma-Chlordane	Arsenic			
Dieldrin	Chromium			
Endrin	Cobalt			
Endrin aldehyde	Iron			
Endrin ketone	Lead			
Heptachlor	Manganese			
Heptachlor epoxide	Mercury			
Toxaphene	Vanadium			
Σ3PC				

Because of the carcinogenic PAH exceedances of the residential soil RSL in both surface and total soil for EUs 1 and 4, all of the detected carcinogenic PAHs that did not exceed the residential RSL were also selected as COPCs (USEPA, 2000a). These included benzo(k)fluoranthene, chrysene, and indeno(1,2,3-cd)pyrene (EU1 only). Similarly, even though DDD did not exceed the residential soil RSL in surface and total soil for EU1, due to the DDE and DDT exceedances, it was also selected as a surface and total soil COPC. Delta-BHC (surface soil only) and gamma-BHC (surface and total soil) were retained as COPCs in EU1 due to the residential RSL exceedances of alpha-BHC and beta-BHC. Lastly, endrin aldehyde was retained as a surface and total soil COPC in EU1 due to the exceedances of endrin and endrin ketone in both surface and total soil.

The metals were subjected to a background comparison. Table 7-10 presents a summary of the metals detected in the background samples. The comparisons of site surface soil metals concentrations to the background values are shown in Table 7-11. Per USEPA Region 4 guidance (USEPA, 2000a), the site maximum concentrations were compared with two times the background average concentrations. Of the metals with maximum surface soil concentrations greater than the RSLs, the site levels of chromium and iron in EU1; chromium, iron, and vanadium in EU2; chromium, manganese, and vanadium in EU3; and

arsenic in EU4 were less than background. The background comparisons for the remaining metals that exceeded the RSLs indicate that the site levels were greater than the background levels. As mentioned previously, those metals eliminated as COPCs for surface soil based on the background comparison were also eliminated as COPCs for total soil. Thus, the following metals were identified as surface and total soil COPCs in the HHRA:

- EU1 – Aluminum, antimony, arsenic, cobalt, lead, manganese, mercury, and vanadium
- EU2 – Aluminum and arsenic
- EU3 – Aluminum, arsenic, cobalt, and iron
- EU4 – Aluminum

In addition to the selection of COPCs, a separate comparison was performed comparing soil concentrations to Georgia EPD Type 1 Risk Reduction Standards (RRS). This analysis is shown in Appendix III – Appendix A. Table A-1 presents the derivation of the RRS and Table A-2 presents the comparison of site levels with the RRS.

7.2 TOXICITY ASSESSMENT

The toxicity assessment examines information concerning the potential human health effects of exposure to COPCs. The goal of the toxicity assessment is to provide, for each COPC, a quantitative estimate of the relationship between the magnitude and type of exposure and the severity or probability of human health effects. The toxicity values presented in this section are integrated with the information presented in the exposure assessment (Section 7.4) to characterize the potential for the occurrence of adverse health effects.

Cancer slope factors (CSFs) and inhalation unit risk factors (URFs) are the dose-response values used to evaluate potential carcinogens. Noncancer effects, such as organ damage or reproductive effects, are evaluated by reference doses (RfDs) and reference concentrations (RfCs). The following hierarchy was used for selection for toxicity values:

- Tier 1 – Integrated Risk Information System (IRIS) (USEPA, 2012a).
- Tier 2 – Values presented on the most recent RSL Table (USEPA, 2012b). Toxicity values presented on the RSL Table are from a number of sources including USEPA (Provisional Peer-Reviewed Toxicity Values), the California Environmental Protection Agency (CalEPA), and the Agency for Toxic Substance and Disease Registry (ATSDR).
- Tier 3 – Derived values.

7.2.1 Non-Cancer Effects

For noncancer effects, it is assumed that there exists a dose below which no adverse health effects would occur. Below this "threshold" dose, exposure to a COPC can be tolerated without adverse effects. Therefore, for noncancer effects, a range of exposure exists that can be tolerated. Toxic effects are manifested only when physiologic protective mechanisms are overcome by exposures to a COPC above its threshold level.

The potential for noncancer health effects resulting from oral or dermal exposure to COPCs is assessed by comparing an exposure estimate (intake or dose) to an RfD. The RfD is expressed in units of mg/kg-day and represents a daily intake of COPC per kilogram of body weight that is not sufficient to cause the threshold effect of concern. An RfD is specific to the COPC, the route of exposure, and the duration over which the exposure occurs.

The potential for non-carcinogenic health effects resulting from inhalation exposure to chemicals is assessed by comparing an exposure concentration in air to an RfC. The RfC is expressed in units of milligrams per cubic meter (mg/m³).

Dermal RfDs are derived from the corresponding oral values. To derive the dermal RfD, the oral RfD (based on an administered dose) is multiplied by the gastrointestinal tract absorption efficiency factor to determine an RfD based on an absorbed dose rather than an administered dose. The resulting dermal RfD is used to evaluate the dermal (absorbed) dose calculated by the dermal exposure algorithms.

Oral RfDs are presented in Table 7-12. Dermal RfDs and the absorption efficiencies used in their determination are also included in Table 7-12. The absorption efficiencies were obtained from USEPA's RAGS Part E Guidance (USEPA, 2004). Inhalation RfCs for the COPCs are presented in Table 7-13. The tables also include the primary target organs affected by each listed COPC, where information is available. This information may be used in the risk characterization to segregate risks by target organ effects when the total hazard index (HI) is greater than 1.0.

A RfD for WT (Σ 3PC) does not exist in either of the Tier 1 or Tier 2 categories listed above and instead was derived from the best available data. The RfD derived for Σ 3PC is based on a no-observed-adverse-effect-level (NOAEL) from an *in vivo* study (MATT, 2000) of rats exposed to WT (Simon and Manning, 2006). Technical toxaphene was weathered for 2 months in farmed codfish and the WT (including primary congeners p-25, p-50, and p-62) was extracted from the cod liver. The WT was then administered to female Sprague-Dawley rats via weekly subcutaneous injections for 20 weeks. At the end of the 20 weeks, the rat livers were examined and showed evidence of the liver as the critical endpoint. The highest dose of 4.17 mg/kg-week was below the threshold for the critical effect and was used to derive a NOAEL of 0.002 mg/kg-day for Σ 3PC. An uncertainty factor of 100 was applied to the

NOAEL to account for an animal-to-human extrapolation, as well as human variability to arrive at a RfD of 2E-05 mg/kg-day for Σ3PC.

7.2.2 Cancer Effects

The toxicity information considered in the assessment of potential carcinogenic risks includes slope factors and a weight-of-evidence narrative consistent with USEPA's 2005 Guidelines for Carcinogenic Risk Assessment (USEPA, 2005). These guidelines use standard narrative descriptors (Carcinogenic to Humans, Likely to Be Carcinogenic to Humans, Suggestive Evidence of Carcinogenic Potential, Inadequate Information to Assess Carcinogenic Potential, and Not Likely to Be Carcinogenic to Humans) to describe the likelihood that a COPC is a human carcinogen and are based on an evaluation of the available data from human and animal studies.

The CSF is the toxicity value used to quantitatively express the carcinogenic hazard of cancer-causing COPCs via oral and dermal routes. It is defined in the IRIS glossary as: "An upper-bound, approximately a 95 percent confidence limit, on the increased cancer risk from a lifetime exposure to an agent. This estimate, usually expressed in units of proportion (of a population) affected per mg/kg-day, is generally reserved for use in the low-dose region of the dose-response relationship, that is, for exposures corresponding to risks less than 1 in 100."

Inhalation risks are determined through use of inhalation URFs (USEPA, 2009a). An inhalation URF is an upper-bound excess lifetime cancer risk estimated to result from continuous exposure to an agent at a concentration of 1 microgram per cubic meter ($\mu\text{g}/\text{m}^3$) in air.

Dermal CSFs are derived from the corresponding oral values. To derive the dermal CSF, the oral CSF is divided by the gastrointestinal absorption efficiency factor to determine a CSF based on an absorbed dose rather than an administered dose.

Oral CSFs are presented in Table 7-14. Dermal CSFs and the absorption efficiencies used in their determination are also included in Table 7-14. Inhalation URFs for the COPCs are presented in Table 7-15.

For carcinogens that act with a mutagenic mode of action for carcinogenesis, USEPA recommends application of Age-Dependent Adjustment Factors (ADAFs) to the cancer slope factor to address early lifetime exposures and the increased susceptibility of children to carcinogens (USEPA, 2005). Further discussion of ADAFs is provided in Section 7.4.1.

7.2.3 Lead

USEPA has not assigned verified or provisional toxicity values (i.e., CSFs and RfDs) to lead. Therefore, lead exposure and risk was not evaluated using the conventional risk assessment approach. USEPA's Integrated Exposure Uptake Biokinetic (IEUBK) Model and Adult Lead Model (ALM) were used to characterize lead risk to individuals based on environmental exposure to lead-contaminated media. Sections 7.4.3 and 7.4.4.5 discuss the evaluations for lead exposure in greater detail.

7.3 EXPOSURE ASSESSMENT

The objective of the exposure assessment is to estimate the nature, extent, and magnitude of potential exposure of humans to COPCs considering both current and future land uses. The exposure assessment involves several steps, which are listed below:

- Evaluating the exposure setting, including describing current and future land and water uses and identifying potentially exposed human populations.
- Developing the conceptual site model including sources, release mechanisms, transport and receiving media, exposure media, exposure scenarios, exposure routes, and potentially exposed populations.
- Calculating exposure point concentrations (EPCs) for each of the exposure scenarios and routes of exposure.
- Identifying the exposure scenarios, models, and parameters with which to calculate exposure doses.

To provide a range of exposure and risks, the reasonable maximum exposure (RME) and central tendency exposure (CTE) scenarios were evaluated (USEPA, 1992). The RME, an estimate of the high-end exposure in a population, is based on a combination of average and high-end estimates of exposure parameters typically representing the 90th percentile or greater of actual expected exposure. The CTE represents an estimate of the average exposure in a population and is based on central estimates of exposure parameters. Both the RME and CTE were evaluated for each exposure scenario.

7.3.1 Exposure Setting

7.3.1.1 Current and Future Land Uses

The HHRA evaluated potential risks associated with the current and reasonably anticipated future uses of OU5. Current land uses, based on Peach County zoning classifications (Figure 7-1), formed the basis for

the evaluation of existing (i.e., baseline) conditions. Future land uses formed the basis for the evaluation of risks associated with future use of OU5.

Current land uses near the former facility consist of industrial, commercial, and residential areas. The majority of the former facility consists of buildings, storage tanks, and stockpiles of equipment. Land use varies along the OU5 drainage ditch and includes industrial and commercial facilities and rural agriculture. A wetland, as well as rural, medium-density, and mobile home residences also lie in close proximity to OU5.

Future plans regarding land use in OU5 are important to the identification of reasonably anticipated future uses of OU5 and the potential for future contact with contaminated media. The future land use is likely to be similar to the current land use because the drainage ditch area is unlikely to be developed for use due to its propensity for flooding and close proximity to wetlands. However, to provide the most conservative assessment, future residential exposure is evaluated in this risk assessment.

7.3.2 Conceptual Site Model

A conceptual site model (CSM) describes the contaminant sources, the release and transport mechanisms, the receiving media, the exposure media, the exposure routes, and the potentially exposed populations. The primary objective of the CSM is to identify complete and incomplete exposure pathways. A complete exposure pathway has all of the above-listed components; whereas, an incomplete pathway is missing one or more. Figure 7-6 illustrates the CSM that was developed for OU5. Each component of the CSM is examined in detail in the following sections.

7.3.2.1 Source of Contamination, Release/Transport Mechanisms, and Receiving Media

COPCs released in the past from the site have been transported from the source area to the downgradient drainage ditch that constitutes OU5. The following potential contaminant transport pathways have been identified:

- Surface runoff and drainage from the site.
- Wind erosion and downstream transport of contaminated sediment and soil.
- Sediment contamination via runoff carrying suspended soil particles.
- Soil contamination via deposition of suspended sediment during flood events.
- Erosion of contaminated soil (surface and subsurface) during flood events, and subsequent deposition as contaminated sediment.

7.3.2.2 Primary Exposure Media

Based on the review of the current and potential land uses, the following primary exposure media are of potential concern to humans in OU5:

- Soil
- Sediment

7.3.3 Characterization of Potentially Exposed Populations

Potential exposure of individuals who trespass and/or recreate along the drainage ditch was examined. Recreational exposure is the most likely exposure occurring within OU5. The recreational users were assumed to contact the surface soil (0 to 0.5 foot bgs) through the incidental ingestion, dermal contact and absorption, and inhalation of particulate exposure routes. Adolescent and adult receptors were evaluated. Adolescents (7 through 16 years) and adults were assumed to be the most frequently exposed receptors based on the nature of the area and the difficulty a young child would likely experience attempting to recreate in the floodplain area. The young child (1 through 6 years) receptor was considered for evaluation, but it was determined that the OU5 area is not conducive for activities for young children.

Site groundskeepers may be exposed to contaminants in surface soil (0 to 0.5 foot bgs) via incidental ingestion, dermal contact, and inhalation of particulates while performing routine activities, such as mowing lawns, grounds upkeep and maintenance, and working in and along the drainage ditches. It is assumed that the groundskeeper exposure is occurring at the present time and will continue to occur in the future.

While it is unclear whether any buried utility lines exist in OU5, a site utility worker is evaluated for potential exposure to total soil (0 to 10 feet bgs) in the future through incidental ingestion, dermal contact, and inhalation of particulates during utility installation and maintenance within OU5.

A construction worker scenario was not considered to be a complete exposure scenario because flooding events preclude major construction in the floodplain. Although it is unlikely that a residential exposure pathway would occur at this OU given the potential for flooding, a residential scenario was included to provide an upper-bound estimate of potential site risks.

7.3.4 Exposure Point Concentrations

The subsections below present the methods used to calculate the EPCs using USEPA's ProUCL software program, Version 4.1.00 (USEPA, 2011). The list below presents the process for determining the EPCs.

- If only one or two samples were collected within a data grouping, the EPC is the maximum detected concentration.
- If between three and eight samples were collected within a data grouping, the EPC is the 75th percentile. Full detection limits are used as values for the non-detected samples in these small data sets.
- Similarly, if eight or more samples were collected within a data grouping, but the data set contains fewer than four detects, the EPC is based on an ad-hoc value, i.e., the 75th percentile. As for small data sets, full detection limits are used as values for the non-detected samples.
- If eight or more samples were collected within a data grouping and the data set contains at least four detects, but the data set contains less than 50 percent detects, a nonparametric-based upper confidence limit (UCL)/EPC is considered. The nonparametric-based value is derived using either Kaplan-Meier (KM) or bootstrapping estimation procedures, unless there are fewer than ten detects. If there are fewer than ten detects, the bootstrapping estimates are not considered.
- If eight or more samples were collected within a data grouping and the data set contains at least 50 percent detects, the appropriate distribution of the data set are determined and UCLs/EPCs are selected as guided by the ProUCL supporting documentation. If the recommended UCL exceeds the maximum detected concentration, a Chebyshev-based UCL is selected as the EPC if possible. If the Chebyshev-based UCL is still higher than maximum detected concentration, the maximum concentration is selected as the EPC.

ProUCL calculates 95 percent UCLs using 15 different computation methods, five parametric and ten non-parametric. Parametric methods rely on the estimation of parameters (such as the mean or the standard deviation) describing the distribution of the variable of interest in the population; non-parametric methods do not. The five parametric UCL computation methods include:

- Student's-t UCL.
- Approximate gamma UCL using chi-square approximation.
- Adjusted gamma UCL (adjusted for level significance).
- Land's H-UCL.
- Chebyshev inequality based UCL (using Minimum Variance Un-biased Estimators (MVUEs) of parameters of a lognormal distribution).

The ten non-parametric methods included in ProUCL are:

- The central limit theorem (CLT) based UCL.
- Modified-t statistic (adjusted for skewness) based UCL.
- Adjusted-CLT (adjusted for skewness) based UCL.

- Chebyshev inequality based UCL (using sample mean and sample standard deviation).
- Jackknife method based UCL.
- UCL based upon standard bootstrap.
- UCL based upon percentile bootstrap.
- UCL based upon bias-corrected accelerated (BCA) bootstrap.
- UCL based upon bootstrap-t.
- UCL based upon Hall's bootstrap.

Support documentation (ProUCL outputs) for the calculation of the UCLs is presented in Appendix III – Appendix B. The surface and total soil EPCs used in this HHRA for EUs 1 through 4 are presented in Tables 7-16 through 7-23, respectively.

7.3.5 Exposure Parameters

This section presents the exposure parameters that were used to quantify exposure in terms of contaminant intake (exposure dose). Table 7-24 presents the exposure parameters for trespassers, groundskeepers, and utility workers. Table 7-25 presented the residential exposure parameters. The formulas used in estimating exposure intakes are also shown on the tables.

The absorption fraction (ABS) term (unitless) represents the fraction of COPCs that are assumed to penetrate the skin following dermal contact with contaminated soil. The ABS values for COPCs were obtained from USEPA RAGS Part E guidance (USEPA, 2004) and are presented in Table 7-26. The ABS values were used in both the RME and CTE evaluations.

7.3.5.1 Trespasser Exposure Parameters

Site trespassers and/or recreational users are potentially exposed to COPCs in surface soil through incidental ingestion, dermal contact and absorption, and inhalation of particulates. The trespasser receptors included an adolescent and adult that may trespass on or use the site for various recreational activities.

The average body weight (BW) values for the adolescent (7 through 16 years) and the adult were 45 kilograms (kg) and 70 kg, respectively (USEPA, 1989, 2000a).

The exposure duration (ED) values used in the RME evaluation for the adolescent and adult trespassers were 10 years (calculated based on receptors age) and 30 years, respectively. The adult ED is consistent with USEPA's default residential ED (USEPA, 1997b). The cancer-based averaging time (AT) was based on a 70-year lifetime for both age groups and equates to 25,550 days (70 years x 365 days/year) (USEPA, 1989). The noncancer AT was based on the ED in years multiplied by 365 days/year (adolescent – 3,650 days; adult – 10,950 days).

The particulate emission factor (PEF), which relates the concentration of a COPC in soil to the concentration of dust particles in air, was assumed to be the default value of 5.7E+09 cubic meters per kilogram (m^3/kg) for the Atlanta region (USEPA, 2002b).

RME

Both the adolescent and adult trespassers exposure frequencies (EFs) were assumed to be 104 days/year which assumes exposure two days per week over the course of a year (52 weeks). Both the adolescent and adult trespassers exposure time (ET) on site was assumed to be 2 hours per day.

The incidental soil ingestion rate (IRS) for residential exposure of an adolescent and adult (100 mg/day) was used in the RME evaluation for site trespassers (USEPA, 1991, 1997b). For soil ingestion, a fraction ingested (FI) value of 1.0 was used. This assumes that the exposed individual receives 100 percent of their daily soil intake while trespassing on the site.

The following exposed skin surface area (SA) values were used in the RME evaluation:

- Adolescent – exposed skin surface includes head, hands, forearms, and lower legs. This equates to a SA value of 5,300 centimeters squared (cm^2) (USEPA, 2004).
- Adult – exposed skin surface includes head, hands, and forearms. This equates to a SA value of 3,300 cm^2 (USEPA, 2004).

The following soil-to-skin adherence factor (AF) values were used in the RME evaluation:

- Adolescent – a value of 0.4 milligram per centimeters squared (mg/cm^2) was used, which is the 95th percentile value for children playing in dry soil activity (USEPA, 2004).
- Adult – a value of 0.1 mg/cm^2 was used, which is the 95th percentile value for the commercial/industrial groundskeeper activity (USEPA, 2004).

CTE

Both the adolescent and adult trespassers EFs were assumed to be half of the RME scenario at 52 days/year which assumes exposure one day per week over the course of a year (52 weeks). Both the adolescent and adult trespassers ET on site was assumed to be half of the RME evaluation at 1 hour per day.

The RME parameters for SA were also used for the CTE analysis and are not repeated in this subsection.

An IRS of 50 mg/day was assumed for the adolescent and adult in the CTE evaluation for site trespassers (USEPA, 1991, 1997b). An FI value of 0.5 was used, which assumes that the exposed individual receives 50 percent of their daily soil intake while trespassing on the site.

The following soil-to-skin AF values were used in the CTE evaluation:

- Adolescent – a value of 0.04 mg/cm² was used, which is the geometric mean value for the children playing in dry soil activity (USEPA, 2004).
- Adult – a value of 0.02 mg/cm² was used, which is the geometric mean value for the commercial/industrial groundskeeper activity (USEPA, 2004).

7.3.5.2 Groundskeeper Exposure Parameters

Groundskeepers may be exposed to COPCs in surface soil within OU5 via the incidental soil ingestion, dermal contact and absorption, and inhalation of particulate routes of exposure during routine activities in the drainage area.

The average BW value for the groundskeeper was 70 kg (USEPA, 1989). The PEF was assumed to be the default value of 5.7E+09 m³/kg for the Atlanta region (USEPA, 2002b).

RME

The ED value used in the RME evaluation for the groundskeeper was 25 years (USEPA, 2002b). The groundskeeper EF was assumed to be 52 days/year which assumes exposure one day per week over the course of a year (52 weeks) and the groundskeeper is assumed to spend 8 hours per day on site. The cancer-based AT was based on a 70-year lifetime and equates to 25,550 days (70 years x 365 days/year) (USEPA, 1989). The noncancer AT was based on the ED in years multiplied by 365 days/year (9,125 days).

An IRS of 100 mg/day was used in the RME evaluation for the groundskeeper (USEPA, 2002b). For soil ingestion, an FI value of 1.0 was used. This assumes that the exposed individual receives 100 percent of their daily soil intake while working on site.

The exposed skin SA value used in the RME evaluation for the groundskeeper was 3,300 cm² (exposed skin surface includes head, hands, and forearms) (USEPA, 2004). The soil-to-skin AF value used in the RME evaluation for the groundskeeper was 0.2 mg/cm² (which is the geometric mean value for the utility worker/heavy equipment operator) (USEPA, 2004).

CTE

The ED value used in the CTE evaluation for the groundskeeper was 12 years (half of the RME ED). The groundskeeper EF was assumed to be half of the RME scenario at 26 days/year and the groundskeeper is assumed to spend half of the RME evaluation at 4 hours per day on site.

The RME SA of 3,300 cm²/day was also used for the CTE analysis.

An IRS of 50 mg/day was assumed for the CTE evaluation for the groundskeeper (USEPA, 1991, 1997b). An FI value of 0.5 was used, which assumes that the exposed individual receives 50 percent of their daily soil intake while trespassing on the site.

The soil-to-skin AF value used in the CTE evaluation for the groundskeeper was 0.02 mg/cm² (which is the geometric mean for the commercial/industrial groundskeeper) (USEPA, 2004).

7.3.5.3 Utility Worker Exposure Parameters

Utility workers may be exposed to total soil COPCs within OU5 via the incidental soil ingestion, dermal contact and absorption, and inhalation of particulate routes of exposure.

The ED value used in RME and CTE evaluation for the utility worker was 25 years (USEPA, 2002b). The cancer-based AT was based on a 70-year lifetime and equates to 25,550 days (70 years x 365 days/year) (USEPA, 1989). The noncancer AT was based on the ED in years multiplied by 365 days/year (365 days).

The average BW value for the utility worker was 70 kg (USEPA, 1989). The PEF was assumed to be the default value of 5.7E+09 m³/kg for the Atlanta region (USEPA, 2002b).

RME

The RME EF for the utility worker was 10 days/year, which assumes the utility worker maintains easements, and inspects, repairs, and replaces equipment for 8 hours per day on site.

An IRS for construction workers of 330 mg/day was used in the RME evaluation for the utility worker (USEPA, 2002b). For soil ingestion, an FI value of 1.0 was used. This assumes that the exposed individual receives 100 percent of their daily soil intake while working on site.

The exposed skin SA value used in the RME evaluation for the utility worker was 3,300 cm² (exposed skin surface includes head, hands, and forearms) (USEPA, 2004). The soil-to-skin AF value used in the RME evaluation for the utility worker was 0.3 mg/cm² (which is the 95th percentile soil-to-skin AF value for construction workers) (USEPA, 2004).

CTE

The CTE EF for the utility worker was half of the RME value at 5 days/year for 4 hours per day on site, which is half the RME scenario. The RME SA of 3,300 cm²/day was also used for the CTE analysis.

An IRS for outdoor workers of 100 mg/day was assumed for the CTE evaluation for the utility worker (USEPA, 2002). An FI value of 0.5 was used, which assumes that the exposed individual receives 50 percent of their daily soil intake while trespassing on the site.

The soil-to-skin AF value used in the RME evaluation for the groundskeeper was 0.1 mg/cm² (which is the geometric mean for construction workers) (USEPA, 2004).

7.3.5.4 Residential Exposure Parameters

Residential users are potentially exposed to COPCs in total soil through incidental ingestion, dermal contact and absorption, and inhalation of particulates. The residential receptors include individuals (child and adult) who are assumed to live on the site and would contact site soil through a wide variety of typical activities around the home.

The average BW values for the child (1 through 6 years) and the adult were 15 kg and 70 kg, respectively (USEPA, 1989, 1991, and 1997). The residential EF was assumed to be 350 days/year for both the RME and CTE (USEPA, 2000a).

The cancer-based AT was based on a 70-year lifetime for both all groups and equates to 25,550 days (70 years x 365 days/year) (USEPA, 1989). The noncancer AT was based on the ED in years multiplied by 365 days/year. The PEF was assumed to be the default value of 5.7E+09 m³/kg for the Atlanta region (USEPA, 2002b).

RME

The ED value used in the RME evaluation for the resident was 30 years (24 as an adult and 6 as a child) (USEPA, 2000a). Residential exposure IRS values were used: adult – 100 mg/day and child – 200 mg/day (USEPA, 1991, 1997b). For soil ingestion, an FI value of 1.0 was used.

The following exposed skin SA values were used in the RME evaluation:

- Child – exposed skin surface includes head, hands, forearms, lower legs, and feet. This equates to a SA value of 2,800 cm² (USEPA, 2004).
- Adult – exposed skin surface includes head, hands, forearms, and lower legs. This equates to a SA value of 5,700 cm² (USEPA, 2004).

The following soil-to-skin AF values were used in the RME evaluation:

- Child – a value of 0.2 mg/cm² was used, which is the 95th percentile value for daycare children activity (USEPA, 2004).
- Adult – a value of 0.07 mg/cm² was used, which is the geometric mean value for gardeners (USEPA, 2004).

CTE

The RME parameters for IRS and SA were also used for the CTE analysis and are not repeated in this subsection. The CTE ED value used evaluation was 15 years (9 as an adult and 6 as a child). An FI value of 0.5 was used, which assumes that the exposed individual receives 50 percent of their daily soil intake while on the site.

The following soil-to-skin AF values were used in the CTE evaluation:

- Child – a value of 0.04 mg/cm² was used, which is the geometric mean value for the daycare child soil activity (USEPA, 2004).
- Adult – a value of 0.01 mg/cm² was used, which is the geometric mean value for the groundskeeper activity (USEPA, 2004).

7.4 RISK CHARACTERIZATION

The objective of the risk characterization is to integrate the information developed in the exposure assessment and the toxicity assessment into an evaluation of the potential risks associated with exposure to COPCs. Cancer risks were calculated for those COPCs with evidence of carcinogenicity and for which cancer toxicity values were available. Noncancer health effects were evaluated for COPCs (i.e., including carcinogens) for which noncancer toxicity values were available.

7.4.1 Cancer Risk

Potential cancer risks from oral and dermal exposure were calculated by multiplying the estimated lifetime average daily dose intake that was calculated for a COPC through an exposure route by the exposure route-specific CSF, as follows:

$$\text{Risk} = \text{LADD} * \text{CSF}$$

Where:

LADD = Lifetime average daily dose; intake averaged over a 70-year lifetime as mg COPC/kg-body weight per day.

CSF = COPC- and route-specific cancer slope factor (mg/kg-day)⁻¹.

Potential cancer risks from inhalation exposure were calculated by multiplying the calculated air concentration and the URF as follows:

$$\text{Risk} = \text{CA} * \text{URF}$$

Where:

CA = Air concentration (µg/m³).

URF = Unit risk factor (µg/m³)⁻¹.

As presented in the RSL Table (USEPA, 2012b), the following COPCs exhibit a mutagenic mode of action for carcinogenesis:

Benzo(a)anthracene	Chrysene
Benzo(a)pyrene	Dibenz(a,h)anthracene
Benzo(b)fluoranthene	Indeno(1,2,3-cd)pyrene
Benzo(k)fluoranthene	

For the carcinogens that act with a mutagenic mode of action for carcinogenesis, the potential cancer risks were calculated by applying ADAFs (USEPA, 2005). Two receptors required the application of ADAFs: the adolescent trespasser (7-16 years) and residents. An ADAF of 3 was applied to adolescent trespasser carcinogenic intakes in order to account for the mutagenic mode of action. Appendix III – Appendix C presents the cancer risk calculations for residential exposure including the application of the ADAFs.

USEPA's cancer risk range is an increased risk of developing cancer, based on a plausible upper-bound estimate of risk, of approximately 1 in 1,000,000 (1E-06) to 1 in 10,000 (1E-04).

7.4.2 Noncancer Health Effects

Potential noncancer health effects from oral and dermal exposure were evaluated by the calculation of HQs and HIs. An HQ is the ratio of the average daily dose through a given exposure route to the COPC-specific RfD. The HQ-RfD relationship is illustrated by the following equation:

$$HQ = ADD/RfD$$

Where:

ADD = Average daily dose; estimated daily intake averaged over the exposure duration (mg/kg-day).

RfD = Reference dose (mg/kg-day).

The HQ for the inhalation pathway was calculated as follows:

$$HQ = CA/RfC$$

Where:

CA = Air concentration ($\mu\text{g}/\text{m}^3$).

RfC = Reference concentration (mg/m^3).

HQs were summed to calculate HIs for each scenario. HIs were calculated for each exposure route, and a total HI was calculated based on exposure to the COPCs from exposure routes for each receptor. HIs of less than 1.0 indicate that adverse health effects associated with the exposure scenario are unlikely to occur.

7.4.3 Lead Evaluation

Lead was not a COPC in EUs 2 through 4; therefore, blood lead levels were estimated for exposure to soil using the IEUBK Model and ALM in EU1 only. The models were run using site-specific inputs for soil ingestion and exposure frequency. The remaining model inputs were default values recommended by USEPA. The average lead concentrations were used for the ALM evaluation for soil. The predicted blood lead levels were compared to the target range of 5 to 10 micrograms per deciliter ($\mu\text{g}/\text{dL}$). Appendix III – Appendix D presents the output of the model runs.

7.4.4 Risk Results

Tables 7-27 and 7-28 present a summary of the total RME and CTE cancer risks and noncancer HIs for EUs 1 through 4. Given the number of EUs, exposure scenarios, exposure pathways, and the resulting

number of risk results, the cancer risk results are also presented graphically on Figure 7-7 to provide an overview of risks for each EU. This figure shows the range of estimated cancer risks within each EU and how the range of risks compares with the USEPA acceptable risk range of $1\text{E-}06$ to $1\text{E-}04$. The ranges of noncancer HIs are presented on Figure 7-8 and are compared to a benchmark HI of 1.0. As shown on Figure 7-7, with the exception of the residential scenario, all of the RME and CTE cancer risks fall within or below the acceptable risk range of $1\text{E-}06$ to $1\text{E-}04$. The same situation exists for the noncancer health effects (see Figure 7-8). The following subsections present the cancer risks and noncancer HIs by EU. Appendix III – Appendix E presents the RAGS Part D Tables 7, 9, and 10 for both the RME and CTE evaluations.

7.4.4.1 EU1

The total RME cancer risks for the adolescent and adult trespassers at EU1 were within the USEPA acceptable cancer risk range of $1\text{E-}06$ to $1\text{E-}04$ with total cancer risks of $6.4\text{E-}05$ and $7.2\text{E-}05$, respectively. The total RME cancer risk for the groundskeeper ($3.4\text{E-}05$) and utility worker ($1.9\text{E-}05$) receptors were also within the USEPA acceptable cancer risk range at EU1. The age-adjusted RME residential cancer risk ($6.5\text{E-}04$) exceeded the upper-end of USEPA's risk range.

The total CTE cancer risk for the adolescent and adult trespassers ($5.7\text{E-}06$ and $8.8\text{E-}06$, respectively), the groundskeeper ($1.8\text{E-}06$), and the utility worker ($3.2\text{E-}06$) at EU1 were all within the USEPA acceptable cancer risk range. The age-adjusted CTE residential cancer risk was $2.4\text{E-}04$.

The total RME noncancer HIs for the adolescent and adult trespassers (0.96 and 0.39, respectively), the groundskeeper (0.22) and the utility worker (0.12) were all less than the noncancer benchmark, with all CTE HIs significantly lower. The residential HIs exceeded 1.0 for the RME child (11.5) and adult (1.3) and the CTE child (5.4) evaluations. The majority of the cancer risks and HIs to receptors at EU1 are attributable to arsenic.

7.4.4.2 EU2

The total RME cancer risk for the adolescent and adult trespassers at EU2 were within the USEPA acceptable cancer risk range of $1\text{E-}06$ to $1\text{E-}04$ with total cancer risks of $1.1\text{E-}05$ and $1.5\text{E-}05$, respectively. The total RME cancer risk for the groundskeeper ($6.6\text{E-}06$) and utility worker ($4.2\text{E-}06$) receptors were at the low end of the USEPA acceptable cancer risk range at EU2. The RME residential cancer risk ($1.4\text{E-}04$) slightly exceeded the USEPA risk range.

The total CTE cancer risks for the adolescent and adult trespassers ($1.1\text{E-}06$ and $1.8\text{E-}06$, respectively), the groundskeeper ($3.6\text{E-}07$), the utility worker ($6.9\text{E-}07$), and the residents ($5.4\text{E-}05$) at EU2 were either within or below the USEPA acceptable cancer risk range

The total RME noncancer HIs for the adolescent and adult trespassers (0.19 and 0.082, respectively), the groundskeeper (0.044) and the utility worker (0.028) were all less than the noncancer benchmark, with all CTE HIs significantly lower. The residential HIs exceeded 1.0 for the RME child (2.8) and the CTE child (1.3) evaluations, with both adult HIs below 1.0. The majority of the cancer risks and HIs to receptors at EU2 are attributable to arsenic.

7.4.4.3 EU3

The total RME cancer risks for the adolescent and adult trespassers were within the USEPA acceptable cancer risk range of $1\text{E-}06$ to $1\text{E-}04$ with total cancer risks of $4.1\text{E-}05$ and $5.3\text{E-}05$, respectively. The total RME cancer risk for the groundskeeper ($2.4\text{E-}05$) and utility worker ($1.1\text{E-}05$) receptors were also within the USEPA acceptable cancer risk range. The age-adjusted RME residential cancer risk ($3.8\text{E-}04$) exceeded the USEPA risk range.

The total CTE cancer risks for the adolescent and adult trespassers ($3.9\text{E-}06$ and $6.5\text{E-}06$, respectively), the groundskeeper ($1.3\text{E-}06$), and the utility worker ($1.9\text{E-}06$) were within the USEPA acceptable cancer risk range. The age-adjusted CTE residential cancer risk was $1.5\text{E-}04$.

The total RME noncancer HIs for the adolescent and adult trespassers (0.69 and 0.31, respectively), the groundskeeper (0.17) and the utility worker (0.078) were all less than the noncancer benchmark, with all CTE HIs significantly lower. The residential HIs exceeded 1.0 for the RME child (7.6) and the CTE child (3.7) evaluations, with both adult HIs below 1.0.

The majority of the cancer risks and HIs to receptors at EU3 are attributable to arsenic.

7.4.4.4 EU4

The total RME cancer risks for the adolescent and adult trespassers were within the USEPA acceptable cancer risk range of $1\text{E-}06$ to $1\text{E-}04$ with total cancer risks of $6.5\text{E-}06$ and $1.6\text{E-}06$, respectively. The total RME cancer risk for the groundskeeper ($8.6\text{E-}07$) and utility worker ($4.1\text{E-}07$) receptors were below the USEPA acceptable cancer risk range. The age-adjusted RME residential cancer risk ($5.9\text{E-}05$) was also within the USEPA risk range.

The total CTE cancer risks for the adolescent and adult trespassers ($4.6\text{E-}07$ and $1.9\text{E-}07$, respectively), the groundskeeper ($1.2\text{E-}07$), and the utility worker ($8.3\text{E-}08$) were all below the USEPA acceptable cancer risk range. The age-adjusted CTE residential cancer risk was $2.2\text{E-}05$, within the USEPA risk range. The total RME and CTE noncancer HIs were all below 1.0. The majority of the cancer risks and HIs to receptors at EU4 are attributable to arsenic.

7.4.4.5 Lead

Blood lead levels were estimated for RME adult exposure to soil in EU1 only:

EU1	Average Lead Concentration (mg/kg)	IEUBK Geometric Mean Blood Lead Level (µg/dL)	ALM Geometric Mean Blood Lead Level (µg/dL)
Surface Soil-Trespasser	237	---	1.3
Surface Soil-Groundskeeper	237	---	1.2
Total Soil-Utility Worker	180	---	1.1
Total Soil-Child Resident	180	2.5	---

As shown in the table above, the lead models (IEUBK and ALM) indicate that the receptors at EU1 are not at risk for behavioral neurotoxicity due to exposure to lead in soil (i.e., predicted blood lead levels were less than 5 µg/dL).

7.5 UNCERTAINTY ANALYSIS SUMMARY

The goal of an uncertainty analysis in a risk assessment is to provide to the appropriate decision makers (i.e., risk managers) information about the key assumptions, their inherent uncertainty and variability, and the impact of this uncertainty and variability on the estimates of risk. The uncertainty analysis shows that risks are relative in nature and do not represent an absolute quantification. The subsections that follow identify the major uncertainties inherent in the HHRA process by report section to determine if the calculated risks may have been overestimated or underestimated, and the approximate degree to which this may have occurred.

7.6 UNCERTAINTY ANALYSIS

This section presents a summary of the uncertainties inherent in the risk assessment and includes a discussion of how they may affect the risk estimates and conclusions of the risk analysis. Uncertainty in the selection of COPCs was related to the nature of the database. Conservatism was applied to the selection of COPCs to ensure that no chemical was overlooked. Evaluating all positively detected chemicals may have overestimated risk, but had no significant impact on the conclusions of the analysis. Uncertainty associated with the exposure assessment included the values used as input variables for a given intake route or scenario and the assumptions made to determine EPCs. Use of maximum concentrations because of limited data within each exposure unit may have overestimated risk, but had

no significant impact on the conclusions of the analysis. Use of the default exposure assumptions for an RME residential or industrial receptor provided estimates of risk within the typical range estimated in risk assessments and ensured adequate protection of human health. Uncertainty in the toxicity assessment included the quality of the existing toxicity data needed to support dose-response relationships and the weight-of-evidence used to determine the carcinogenicity of COPCs. The application of uncertainty and modifying factors to the data of toxicological studies used to form the basis of the development of the toxicity factors provides a level of conservatism to ensure protection of human health. Uncertainty in risk characterization was associated with exposure to multiple chemicals and the cumulative uncertainty from combining conservative assumption made in earlier steps of the risk assessment process. The interaction between chemicals may result in synergistic or antagonistic effects, thus resulting in potential overestimates or underestimates of total risk.

7.6.1 Hazard Identification

- **Use of XRF data in the HHRA** – As discussed in Section 7.1.1, XRF data was included in the HHRA for arsenic at locations where there was no fixed-base laboratory data. This adds uncertainty to the overall results of the HHRA, but it is likely to be minor given the strong correlation noted between fixed-base laboratory and XRF data.
- **Analytes without screening values** – A number of detected analytes did not have screening values available and were not carried through the risk assessment process. Because toxicity criteria were not available for these analytes (as demonstrated by a lack of health-based screening concentrations), risks (cancer and noncancer) could not be estimated. It is possible that site risks are slightly underestimated as a result of this but the degree to which they are underestimated cannot be determined.

7.6.2 Exposure Assessment

- **The selection of exposure scenarios** – It is likely that the scenarios evaluated overstated realistic exposures, and thus overestimated the actual site risks. For example, the evaluation of the residential scenario, when it is unlikely that residences would be built near the drainage ditch in OU5, would result in an overestimation of exposure and risk.
- **The selection of exposure assumptions** – The exposure assumptions directly influence the calculated doses (chronic daily intakes), and ultimately the calculation of risk. The RME concept was used to estimate the exposure potential for each of the receptors that were evaluated in the HHRA. The RME is defined as the "maximum exposure that is reasonably expected to occur at the site" (USEPA, 1989). In most cases, these assumptions contribute to an overestimation of plausible real-life exposures, and a resulting overestimation of risk.

- **Calculation of 95 percent UCLs** – As presented in Section 7.3.4, one-side 95 percent UCLs were calculated and used as the EPC. A conservative approach of using the full detection limit for NDs was followed for all COPCs in this HHRA. The resulting value represents a conservative estimate of the COPC concentration to which an individual could be exposed in any given EU during the defined exposure duration and frequency. It is likely that using the full detection limit overestimates the site risk to some degree.

7.6.3 Toxicity Assessment

- **The use of cancer slope factors and reference doses** – Both cancer risks and noncancer health effects were evaluated using USEPA-approved or provisional toxicity criteria. The CSFs and RfDs are derived to be health protective and tend to overestimate true toxicity in humans. Therefore, risk calculations, which are partially based on toxicity estimates, may be overstated in general. The exact degree of overestimation cannot always be determined and each COPC must be evaluated on a case-by-case basis.
- **Lack of toxicity values for dermal exposure** – Toxicity values for dermal exposures have not been developed by USEPA. Oral RfD and oral CSFs were adjusted and used to assess toxicity from dermal exposures following guidelines provided by the USEPA. The dermal route of exposure can result in different patterns of distribution, metabolism, and excretion than occur from the oral route. When oral toxicity values for systemic effects are applied to dermal exposures, uncertainty in the risk assessment is introduced because these differences are not taken into account. Since any toxicity differences between oral and dermal exposure would depend on the specific COPC, use of oral toxicity factors can result in the overestimation or underestimation of risk. It is not possible to make a general statement about the direction or magnitude of this uncertainty.
- **Dermal carcinogenicity of PAHs** – The majority of animal and human studies of PAH exposure strongly suggest that the carcinogenic effects resulting from exposure occur at the site of contact or administration (e.g., skin tumors from dermal contact, GI tumors from oral contact) (ATSDR, 1995). There is little evidence that PAHs produce systemic tumors following dermal contact (ATSDR, 1995). In order to justify the extrapolation of an oral CSF to a dermal CSF, an assumption must be made that the type of cancer produced by oral administration is the same as that which would be expected following dermal contact (i.e., that dermal contact with PAHs would produce gastrointestinal [GI] tumors). Since this is not believed to be the case, even though dermal absorption has been quantified for PAHs, extrapolation of the oral CSF to the dermal route of exposure introduces a high level of uncertainty into the analysis. Although it is unlikely that GI tumors would be produced by dermal contact with PAHs, since there is evidence that dermal contact with PAHs may cause skin cancer, the only available data (i.e., the oral CSF) was used to quantify potential cancer risk from

dermal contact with PAHs. This approach introduces a high degree of uncertainty into the analysis, and may overestimate the dermal cancer risks from PAHs to a significant degree.

7.7 RISK SUMMARY

With the exception of the residential exposure scenario, the RME and CTE cancer risks for the trespassers, groundskeeper, and utility worker receptors were within or below the USEPA risk range and all of the noncancer HIs were less than the noncancer benchmark of 1.0 (see Tables 7-27 and 7-28 and Figures 7-7 and 7-8). The residential risks exceeded USEPA's applicable cancer and noncancer risk thresholds, but it is unlikely that residential exposure would ever occur in OU5. The residential scenario was evaluated to provide an upper-bound on the potential levels of risk. Given that the overall approach to the HHRA would tend to overestimate actual risks to a fairly significant degree, it is unlikely that exposure to OU5 site soils would result in any unacceptable health impacts for the evaluated non-residential receptors.

8.0 ECOLOGICAL RISK ASSESSMENT

8.1 INTRODUCTION

This final revised Risk Characterization (Step 7) is an update to the April 2009 update to the June 2008 Step 7 report (ILS, 2008). This revision differs from the earlier versions of Step 7 in that it includes additional chemistry and toxicity data. Additional data collected include:

- 253 soil/sediment samples;
- 6 invertebrate samples;
- Toxicity testing for *Hyalella azteca*, *Eisenia foetida*, and *Lumbriculus variegatus*; and
- Bioaccumulation testing for *E. foetida* and *L. variegatus*.

Other revisions were made and are discussed in the appropriate sections below.

This final revised risk characterization (Step 7), along with its accompanying documents, completes the ecological risk assessment (ERA) process for the Woolfolk site, Fort Valley, Georgia. The accompanying documents are as follows:

- Preliminary OU4 Ecological Risk Evaluation, (CH₂MHILL 1997);
- Technical Memorandum, Step 3A of the Problem Formulation for the Baseline Ecological Risk Assessment (BERA), (CDM, 2000);
- Problem Formulation, Baseline Ecological Risk Assessment (Step 3), (ILS, 2006);
- Study Design and Data Quality Objectives (DQO) Process (Step 4) (ILS, 2007a);
- Field Investigation and Data Report (Step 6), (USEPA, 2008a); and
- Step 7 – Risk Characterization, Ecological Risk Assessment, (ILS, 2008).

The final Step of the 8-Step ERA process is Risk Management (Step 8), which is the sole responsibility of the site risk manager. This final Step is a distinctly different process from the ERA (USEPA, 1997a).

In all, five OUs have been designated at the site. Step 7 originally pertained only to sediments along the drainage ditch to Spillers Street, which along with the contamination in attic dust and soils comprised OU4. Soil was not considered as an exposure medium for OU4. OU5 was added to this evaluation and addresses contamination along the drainage ditch as it extends from the Spillers Street pipe to beyond the railroad discharge into the upper tributary of Big Indian Creek. However, it was determined during the site investigations that portions of the wetlands were dry and sediment samples from those dry locations were collected and evaluated both as soil and sediment in the risk characterization. The previous site

activities resulted in the contamination of habitats that could be considered aquatic only and aquatic or terrestrial, which is the basis of this ERA.

Risk characterization, which is the final phase of the ERA process, includes two major components: 1) Risk Estimation and 2) Risk Description (USEPA, 1998, 1997a). Risk estimation consists of integrating the exposure profiles with the exposure-effects information and summarizing the associated uncertainties; and the risk description provides information important for interpreting the risk results. Risk characterization is the culmination of the planning, problem formulation, and analysis of predicted or observed adverse ecological effects related to the assessment endpoints. It is also the starting point for risk management considerations and the foundation for regulatory decision-making (USEPA, 1998, 1997a).

For this final revised risk characterization, the same general lines of evidence that were used in the original risk characterization were used to evaluate the risks to ecological receptors that may be present at the site. Analytical data from the September 2007 sampling event used for the original preparation of this risk characterization are provided in Step 6 of the ERA, data from the 2009 and 2011 sampling events are presented Tables 5-1 through 5-4. All of the available data from chemical analyses, toxicity tests, field observations, and field measurements were used to estimate the likelihood that significant ecological effects are occurring or will occur at the Woolfolk site, and to describe the nature, magnitude, and extent of the effects on the designated assessment endpoints (Suter, 1996). The soil, worm, sediment, surface water, and fish sampling locations are shown in Appendix IV.

The COPCs retained for the site media included arsenic, lead, and various pesticides that were produced by the facility. These COPCs were identified in Step 3 of the BERA and were carried through as the COPCs in this risk assessment even though additional chemical groups were analyzed in samples from the most recent data collection effort. The potential effects of not evaluating volatiles, semivolatiles, and polychlorinated biphenyls (PCBs) in this Step 7 are presented in Section 8.7 – Uncertainty Analysis.

Based on the results of the original Step 7 of the BERA, both terrestrial and aquatic receptors were likely to be at risk from exposure to the COPCs at the site. Aquatic habitats of concern identified at the site include wetland habitat and creeks. Aquatic receptors potentially at risk include benthic invertebrates, fish, and benthivorous and piscivorous wildlife species. The critical receptors in the terrestrial habitats (i.e., wetlands and dry creek bed) include invertebrates, and mammalian, and avian wildlife species.

Step 3 – Problem Formulation (ILS, 2006) listed the assessment and measurement endpoints and COPCs to be evaluated in Step 7 – Risk Characterization. Because risk assessment is an iterative process, additional assessment endpoints were included as portions of the wetlands/creek were found to be dry oftentimes and the sediments act as a soil habitat. In addition, other chemicals (e.g., more pesticides) were identified during the site investigation phase.

The assessment endpoints selected for the BERA in Step 7 are listed below:

- 1) Protection of soil invertebrates;
- 2) Protection of insectivorous, omnivorous, and carnivorous mammals;
- 3) Protection of insectivorous, omnivorous, and carnivorous birds;
- 4) Protection of piscivorous mammals;
- 5) Protection of piscivorous birds;
- 6) Protection of benthic macroinvertebrate communities;
- 7) Protection of fish populations; and
- 8) Protection of reptiles and amphibians.

8.1.1 Site Investigations

The BERA identified additional data needs that are required to complete the ERA for the site. The site investigation (Step 6) was performed by USEPA Region 4 personnel and Region 4 Environmental Services Assistance Team (ESAT) contractors to collect and analyze samples to fill the data gaps. Original field work for the site investigations was conducted during the last week in September 2007. Abiotic samples collected during the site investigation included 14 surface soils (i.e., 11 soil samples, 1 duplicate, and 2 background samples), 5 subsurface soils, 10 surface sediments (i.e., 7 site samples, 1 duplicate, and 2 background samples), and 7 surface water samples (i.e., 4 site waters, 1 duplicate, and 2 background samples).

The biotic samples collected were mainly fish, and included bluegill (*Lepomis macrochirus*), dollar sunfish (*Lepomis marginatus*), redbreast sunfish (*Lepomis auritus*), redbfin pickerel (*Esox americanus*), and flier (*Centrarchus macropterus*). The fish species list, sizes, weights and number used in composite samples, and total gram weight used in this risk estimation are provided in the *Field Investigation Data Report* (Step 6) (USEPA, 2008a).

Chemical analyses or other measurements were performed for one or more of the following parameters in the environmental samples: TAL metals, TCL organics, percent moisture, total organic carbon (TOC), percent lipids, length, and weight. Site-specific acute and chronic toxicity tests were performed with the abiotic media samples.

At the request of the USEPA RPM, 25 additional surface soil and sediment samples were collected by USEPA Region 4 personnel in September 2008 (USEPA, 2008b) and analyzed for arsenic via XRF. The samples were oven-dried in the laboratory and analyzed by XRF. XRF data are normally regarded as screening data; therefore, all of the results were flagged as estimates (i.e., J-flagged).

The BERA Step 7 indicated that the site was not completely characterized, so to address these concerns, additional data were collected. Sample locations were selected based on previous data collected and

included known areas of elevated arsenic (i.e., concentrations above 33 mg/kg, the previously established freshwater sediment ecological screening value) (J.M. Waller, 2009b). In August, November, and December 2009, XRF was used to analyze 104 surface sediment samples (plus 1 duplicate) and 105 sediment samples that were 0-0.5 foot bgs (plus 6 duplicates) for arsenic, in addition to numerous subsurface samples. Fixed laboratory analyses were also performed on 36 co-located 0-0.5 foot bgs samples for a suite of metals.

For the invertebrate soil/sediment toxicity tests, samples were collected in April 2010. Two background sediment (0-0.5 foot bgs) samples were collected, as well as four site sediment and two site soil samples (0-0.5 foot bgs) plus one duplicate. These samples were analyzed for volatiles, semivolatiles, pesticides/PCBs, toxaphene parlars, and metals. Chemical analyses were also performed on five invertebrate samples plus one duplicate – two *E. foetida* and three *L. variegatus*. Invertebrates were analyzed for pesticides, toxaphene parlars, and arsenic.

8.1.2 Data Evaluation/Reduction

Surface water and fish data were used as presented in the original Step 7 report.

For this revised Step 7 of the BERA, sediments up to 0.5 foot deep only were used (i.e., subsurface soil results were not used). The 2009 XRF data showed good correlation with the samples analyzed at the fixed laboratory. As such, for samples that had only XRF data, these results were used (i.e., when a sample was analyzed by both XRF and the fixed laboratory, only the fixed laboratory data were used). XRF samples had no detection limits associated with them; therefore, if arsenic was non-detect in an XRF sample, the sample was not used. This occurred in 46 sediment samples (up to 0.5 foot bgs) from 2009. Lastly, if a sample was collected as soil, it was assumed that that location periodically dries out and all samples at that location (regardless of date or condition when sampled) were considered soil and used in the terrestrial evaluation. Because the “soil” samples were collected from within wetlands or the creek drainage/floodplain, they were also considered sediment.

Based on the available benchmarks for PAHs, BHC isomers, chlordane isomers, DDT and metabolites, endosulfans, endrins, heptachlor, and toxaphene parlars, the individual chemical concentrations within a sample were summed to obtain sample-specific concentrations. If individual chemical was a nondetect within a sample, one-half the SQL was used as a surrogate concentration for the summation. Site statistics were calculated using one-half the SQL for non-detected samples.

Note that tentatively identified compounds (TIC) and “R” qualified data were not used in the soil/sediment evaluation or in concentration inputs to the receptor modeling.

Summaries statistics for the site soil, sediment, and invertebrate data used in this revised Step 7, as well as the reference area statistics are presented in Tables 8-1 through 8-6.

8.2 RISK ESTIMATION

Risk estimation integrates the exposure profiles with the exposure-effects information. Two main lines of evidence are presented in the risk characterization: 1) HQs and 2) toxicity test results.

In the risk estimation, HQs were calculated by dividing site contaminant concentrations by benchmark values or estimated average daily intakes by TRVs. An HQ of 1.0 (i.e., unity) or greater indicates that there may be unacceptable risks to ecological receptors from exposure to that chemical. The individual toxicity benchmarks or alternative toxicity values (ATVs) used in the medium-specific evaluations are based on the sources are provided in the BERA (ILS, 2006) and for the original as well as new COPCs, presented within text below. TRVs are presented in Table 8-7.

8.3 CHANGES FROM THE DRAFT RISK CHARACTERIZATION

The assumptions, methodologies, equations, and input parameters used for all of the food-web models are described in the Study Design and DQO Process Work Plan for the Woolfolk site (ILS, 2007a) or the Sampling and Analysis Plan, Quality Assurance Project Plan (ILS, 2007b). However, as needed, EPCs and exposure scenarios were updated. In addition, subsequent to the Risk Characterization report (ILS, 2008), additional data were collected, from which bioaccumulation factors could be calculated for some COPCs. These changes are discussed in the sections below.

8.3.1 Exposure Point Concentrations and Exposure Scenarios

For the medium-specific COPC concentration comparisons to benchmark concentrations, it was determined that it is more appropriate to use a sample-by-sample approach than to compare the benchmarks to just the maximum and average concentrations. For the most part, receptors evaluated using the medium-specific benchmarks are either sessile (as in the case of plants) or have limited mobility (as in the case of invertebrates) and would not be exposed to an “average” site concentration. Comparing on a sample-by-sample basis allows for easier interpretation of results as the range of HQs can be noted, as well as the individual samples that have HQs greater than unity, thereby determining potential locations where cleanup may be warranted.

For food chain modeling, it was determined that two scenarios would be sufficient to capture the range of risks – the maximum and average scenarios. For the maximum scenario, all concentration inputs (e.g., soil/sediment ingestion, water ingestion, prey item ingestion) are, or are based on the maximum detected concentration. For the average scenario, all concentrations are, or are based on the arithmetic average concentration.

8.3.2 Assessment Endpoint No. 1 – Protection of Soil Invertebrates

The risk questions developed for this assessment endpoint include the following:

1. Are the COPC concentrations in the site soils elevated enough in comparison to soil benchmark values and the reference stations to cause a reduction in the survival, growth, and/or reproduction of soil-dwelling invertebrates?
2. Do acute toxicity tests show that COPC concentrations in the site soils are high enough to cause mortality and/or reduction in growth to soil-dwelling invertebrates?

The measurement endpoints for this assessment endpoint are:

1. Comparison soil COPC concentrations to soil benchmark values; and
2. Results of earthworm (*Eisenia foetida*), acute toxicity tests.

8.3.2.1 Comparison of Soil COPC Concentrations to Soil Benchmark Values

Soil benchmark values were used in conjunction with sample-specific COPC concentrations to calculate HQs as estimates of risk to soil invertebrates. The ATVs used in this risk characterization are the higher of the benchmarks presented in the soil summary tables (Tables 8-1 and 8-2) and those presented in the Problem Formulation (Step 3 – ILS, 2006). A summary of the samples that exceed the soil ATVs is presented in Table 8-8 with the individual sample HQs presented in Appendix IV – Appendix B (Table B-1). Note that the frequency of detections in Table 8-8 may be different than those presented in the summary tables (Tables 8-1 and 8-2) as the individual sample HQ calculations treated primary and field duplicate results separately; whereas, the frequency of detection in the summary tables treated primary and field duplicates as one sample.

Arsenic

Arsenic is widespread in the surficial soils at the site and most of the locations exceeded USEPA's ecological soil screening level (Eco-SSL) of 18 mg/kg for the protection of plants. An Eco-SSL for invertebrates is not available (USEPA, 2005a). The ATV selected for arsenic in surface soil was the Dutch intervention value of 55 mg/kg (MHSPE, 2000). The intervention value is a less conservative toxicity value which is expected to be hazardous to 50 percent of all the ecological receptors at a site. Values above the intervention value are regarded as indicative of serious contamination, which require cleanup. The maximum arsenic concentration at the site generated an HQ greater than unity (i.e., station WC2160410 with HQ=27.3). Arsenic concentrations in 26 out of 34 samples exceeded the ATV. The arsenic concentrations from the background stations generated HQs less than unity.

Lead

The ATV selected for lead in soil was 150 mg/kg, which is regarded as a value that represents moderate contamination for all ecological receptors and requires further study (Beyer, 1990). The maximum lead concentration of 1,600 mg/kg generated an HQ of 10.7. In all, 10 out of the 19 surface soil samples generated HQs greater than unity. It must be noted that none of the samples exceeded soil invertebrate-specific Eco-SSL of 1,700 mg/kg for lead (USEPA, 2005b).

Benzene Hexachloride (BHC) Isomers

Various isomers of BHC (i.e., alpha, beta, delta, and gamma) were detected in the surficial soils at the site. Gamma-BHC (Lindane) is the only BHC isomer with pesticidal properties. For the purposes of this risk characterization, the concentrations of each isomer were compared to the TRV of 0.010 mg/kg for hexachlorocyclohexane (HCH) (CCME, 2007). This benchmark value was developed for the protection of human health and the environment.

The isomers were not summed as the gamma isomer is most toxic and assuming that each of the other isomers contributes the same toxicity is overly conservative. All but the delta isomer were detected in approximately half of the soil samples; whereas, delta-BHC was only detected in 2 of 17 samples. BHC isomers exceeded the benchmark as follows:

- alpha-BHC: 6/10 samples – maximum HQ = 15, the rest of the HQs were below 10;
- beta-BHC: 9/9 samples – maximum HQ = 87, the rest of HQs were below 10;
- delta-BHC: 2/3 samples – maximum HQ = 2.9; and
- gamma-BHC: 7/10 samples – maximum HQ = 14, the rest of the HQs were below 10.

BHC isomers were not detected in background.

Chlordane

The chlordane isomers (alpha and gamma) were summed and compared with a USEPA Region 5 ecological screening level (ESL) plant value ATV of 0.224 mg/kg (USEPA, 2003). The maximum calculated total chlordane concentration of 28 mg/kg generated an HQ of 125. Nine out of 12 samples generated HQs greater than unity, seven HQs of which were greater than 10, with one being greater than 100. The chlordanes were not detected in background.

DDT and Metabolites

DDT was used at the facility and was detected in most of the surface soil samples in varying concentrations. DDE, a metabolite of DDT, was also detected in most of the surface soil samples in varying concentrations. The ATV for DDT and metabolites was the Canadian soil quality guideline value

of 0.700 mg/kg (CCME, 2007). This benchmark value was developed for the protection of human health and the environment. The maximum calculated DDT and metabolite concentration of 111.8 mg/kg generated an HQ of 160. Nine out of 16 samples generated HQs greater than unity, three HQs of which were greater than 10, with one being greater than 100. The background DDT concentrations generated HQs less than unity.

Dieldrin

Dieldrin concentrations were compared with a USEPA Eco-SSL for mammals of 0.0049 mg/kg. An Eco-SSL for soil invertebrates was not available (USEPA, 2005c). The maximum concentration of 9.1 mg/kg generated an HQ of 1,860. All 13 samples where dieldrin was detected had HQs greater than unity; three of which were between 10 and 100, and six of which were greater than 100. The dieldrin concentrations in background generated HQs less than unity.

Endosulfan II

Endosulfan II concentrations were compared with a USEPA Region 5 ESL for the masked shrew of 0.119 mg/kg (USEPA, 2003). The maximum concentration of 2.7 mg/kg generated an HQ of 22.7. Eight of nine samples where endosulfan II was detected had HQs greater than unity. Of these, only one sample had an HQ greater than 10. Endosulfan II was not detected in background.

Endosulfan Sulfate

Endosulfan sulfate concentrations were compared with a USEPA Region 5 ESL for the masked shrew of 0.0358 mg/kg (USEPA, 2003). The maximum concentration of 1.1 mg/kg generated an HQ of 15.4. All six of the samples where endosulfan sulfate was detected had HQs greater than unity. Of these, only one sample had an HQ greater than 10. Endosulfan sulfate was not detected in background.

Endrin, Endrin Aldehyde, and Endrin Ketone

Endrin, endrin aldehyde, and endrin ketone concentrations were compared with a USEPA Region 5 ESL for the masked shrew of 0.0101 mg/kg for endrin (USEPA, 2003). These compounds were not summed as their relative toxicity is not well known. Endrin-related compounds were detected in either only 2 or 3 of 16 samples. Endrin-related compounds exceeded the benchmark as follows:

- Endrin: 3/3 samples – maximum HQ = 564, rest above 10;
- Endrin aldehyde: 2/2 samples – maximum HQ = 129, the other above 10; and
- Endrin ketone: 2/2 samples – maximum HQ = 752, the other above 10.

Endrin was not detected in background.

Heptachlor

Heptachlor concentrations were compared with a USEPA Region 5 ESL for the masked shrew ATV of 0.00598 mg/kg for heptachlor (USEPA, 2003). The maximum concentration of 0.51 mg/kg generated an HQ of 853. All four of the samples where heptachlor was detected had HQs greater than unity; three of which had HQs greater than 10. Heptachlor was not detected in background.

Heptachlor Epoxide

Heptachlor epoxide concentrations were compared with a USEPA Region 5 ESL for the masked shrew ATV of 0.152 mg/kg for heptachlor epoxide (USEPA, 2003). The maximum concentration of 0.23 mg/kg generated an HQ of 1.51. Only one of the three samples where heptachlor epoxide was detected had HQs greater than unity. Heptachlor epoxide was not detected in background.

Toxaphene

Toxaphene concentrations were compared with a USEPA Region 5 ESL for the masked shrew ATV of 0.119 mg/kg (USEPA, 2003). The maximum concentration of 170 mg/kg generated an HQ of 1,430. All six samples where toxaphene was detected had HQs greater than 10; two of which were greater than 100. Toxaphene was not detected in background. Toxaphene parlars – sum were also compared with the Region 5 ESL. The maximum concentration of 6.69 mg/kg generated an HQ of 56.2. All four samples where toxaphene was detected had HQs greater than 1.

8.3.2.2 Risk Evaluation Using Site-Specific Surface Soil Toxicity Data

Whole soil toxicity tests using *E. foetida* were used as a measurement endpoint to assess risks to soil invertebrates from exposure to COPCs in site soils. The earthworm toxicity tests were designed primarily to evaluate the direct toxicity of arsenic to the earthworms. Therefore, a range of known arsenic concentrations in sediments were used to select samples included in the tests. Acute earthworm soil toxicity tests were conducted in 2007 (USEPA, 2007) and again in 2010 (USEPA, 2010). In 2010, a 28-day earthworm bioaccumulation study was also conducted using *E. foetida*; however, the bioaccumulation study is not a line of evidence for this assessment endpoint but the results are used to assess risk to insectivorous, omnivorous, and carnivorous birds and mammals. The complete reports for the 2007 and 2010 toxicity tests conducted by USEPA are provided in Appendix IV – Appendix C and D, respectively.

Fourteen day acute toxicity tests were performed using the lumbricid earthworm, *E. foetida*, with surficial soils collected at two stations in 2007 (i.e., WC-106-SS and WC-109-SS) and two stations in 2010 (i.e., WC-175 and WC-216). The locations of the surface soil samples used in the soil toxicity tests are presented in Appendix IV on Figure 7-1. A laboratory control soil (artificial soil) also was included for

each test. For both tests, the artificial soil used as a laboratory control was prepared using 70 percent fine sand, 20 percent kaolinite clay, and 10 percent peat moss.

The toxicity tests were conducted using modifications to procedures from USEPA's *Protocols for Short-term Toxicity Screening of Hazardous Waste Sites* (Greene et al., 1989) and American Society for Testing and Materials (ASTM) guidelines entitled *Standard Guide for Conducting Laboratory Soil Toxicity or Bioaccumulation Tests with the Lumbricid Earthworm; Eisenia foetida* (ASTM, 1997). The specific methods and any modifications used in the tests can be found in the *Field Investigation Data Report – Step 6* (USEPA, 2008a). The endpoints for the *E. foetida* toxicity test were growth and survival. Although growth, as measured by weight change, is not considered a strong endpoint because adult worms are not expected to grow appreciably during the 14-day study, as observed in the test conducted in 2007.

As such, growth was not recorded in the 2010 toxicity tests, but was measured at the end of the 28-day bioaccumulation study; however, due to hydration problems that occurred during the bioaccumulation study, weight loss observed cannot necessarily be attributed as a site-related adverse effect. As with the 2007 study, no adverse survival effects were observed during either of the 14-day toxicity tests for *E. foetida*. The highest contaminant concentrations were observed in sample WC-216, which had a 97.5 survival rate, although worms from this sample appeared lethargic at the end of the 14-day test. Concentrations of some of the contaminants of greatest concern in this sample were as follows: arsenic 1,500 mg/kg, lead 1,600 mg/kg, DDT_r 111,800 µg/kg, and toxaphene 170,000 µg/kg. A summary of the test results are found in Table 8-9.

8.3.2.3 Summary of Risks to Soil Invertebrates

Based on the HQs calculated from the soil concentrations and the soil benchmark values, the following locations generated HQs that were greater than unity and may have soil concentrations high enough to cause adverse effects to soil invertebrates:

- Arsenic – 27 Woolfolk stations as noted in Appendix IV – Appendix B (Table B-1).
- Pesticides – Woolfolk stations 106, 107, 108, 175, 177, and 216. For dieldrin, stations 109, 110, and 111 also have HQs greater than unity.

All of the locations where pesticides had HQs greater than unity, the concentration of arsenic was greater than the benchmark value.

However, 14-day toxicity tests with *E. foetida* indicated no adverse effects on survival at soil concentrations ranging from 54 to 1,500 mg arsenic/kg. Given that, at this site, lead and pesticide concentrations have good positive correlation to arsenic concentrations (see Section 8.4.3.1), and that toxicity tests indicated no adverse effects for survival in the sample with the highest arsenic

concentration, COPC-induced mortality is unlikely in samples with arsenic concentrations less than 1,500 mg/kg.

8.3.3 Assessment Endpoint No. 2 – Protection of Insectivorous, Omnivorous, and Carnivorous Mammals

The risk question developed for this assessment endpoint is:

Are the COPC concentrations in the site soils, surface water, and prey species elevated enough to cause adverse effects to the long-term health and reproductive capacity of predatory mammals?

The measurement endpoint for this assessment endpoint is a food-web model. The line of evidence considered for the evaluation of risks to insectivorous, omnivorous, and carnivorous mammals (hereafter referred to as insectivorous mammals) is HQs from the comparison of estimated daily doses to the short-tailed shrew (*Blarina brevicauda*) to no-observed-adverse-effect level (NOAEL) and lowest-observed-adverse-effect level (LOAEL) based TRV.

HQs for the insectivorous mammal are presented in Tables 8-10 and 8-11 for the maximum and average scenarios, respectively. More details are discussed below.

Arsenic

HQs generated for arsenic exceeded unity for all scenarios. HQs ranged from 4 to 41 (average case LOAEL-based to maximum case NOAEL-based).

Lead

HQs generated for lead exceeded unity for maximum case scenarios and for the average case NOAEL. HQs ranged from 0.11 to 5.8 (average case LOAEL-based to maximum case NOAEL-based).

BHC Isomers Sum

HQs generated for BHC isomers – sum did not exceed unity for any scenario.

Chlordane Isomers – Sum

HQs generated for chlordane exceeded unity for all scenarios. HQs ranged from 15 to 201 (average case LOAEL-based to maximum case NOAEL-based).

DDT and Metabolites – Sum

HQs generated for DDT and metabolites exceeded unity for all scenarios. HQs ranged from 1.8 to 14 (average case LOAEL-based to maximum case NOAEL-based).

Dieldrin

HQs generated for dieldrin exceeded unity for all scenarios. HQs ranged from 15 to 182 (average case LOAEL-based to maximum case NOAEL-based).

Endosulfans – Sum

HQs generated for Endosulfans – sum did not exceed unity for any scenario.

Endrins – Sum

Only the NOAEL-based HQ generated for endrin compounds exceeded unity under the maximum case with an HQ of 1.7.

Heptachlors – Sum

HQs generated for heptachlors – sum exceeded unity for all scenarios. HQs ranged from 1.8 to 17 (average case LOAEL-based to maximum case NOAEL-based).

Toxaphene Parlars – Sum

HQs generated for toxaphene parlars – sum exceeded unity for NOAEL-based scenarios, with HQs ranging from 1.2 to 1.3 (average case and maximum case, respectively).

Toxaphene

HQs generated for toxaphene exceeded unity for all scenarios. HQs ranged from 3.4 to 19 (average case LOAEL-based to maximum case NOAEL-based).

8.3.3.1 Summary of Risks to Insectivorous, Omnivorous, and Carnivorous Mammals

NOAEL-based HQs exceeded unity for arsenic, lead, chlordane isomers – sum, DDT and metabolites – sum, dieldrin, endrins – sum, heptachlors – sum, toxaphene parlars – sum, and toxaphene under the maximum exposure scenario. LOAEL-based HQs exceeded unity for all the same except for endrins – sum and toxaphene parlars – sum.

NOAEL-based HQs exceeded unity for arsenic, lead, chlordane isomers – sum, DDT and metabolites, dieldrin, heptachlors – sum, toxaphene parlars – sum and toxaphene under the average exposure scenario. LOAEL-based HQs exceeded unity for all the same except for lead, and toxaphene parlars – sum.

8.3.4 Assessment Endpoint No. 3 – Protection of Insectivorous, Omnivorous, and Carnivorous Birds

The risk question developed for this assessment endpoint is:

Are the COPC concentrations in the site soils, surface water, and prey species elevated enough to cause adverse effects to the long-term health and reproductive capacity of predatory birds?

The measurement endpoint for this assessment endpoint is a food-web model. The lines of evidence considered for the evaluation of risks to insectivorous, omnivorous and carnivorous birds (hereafter referred to as insectivorous birds) is HQs from the comparison of estimated daily doses to the American woodcock (*Scolopax minor*) to NOAEL- and LOAEL-based TRVs.

HQs for the insectivorous bird are presented in Tables 8-12 and 8-13 for the maximum and average scenarios, respectively. More details are discussed below.

Arsenic

NOAEL- and LOAEL-based HQs generated for arsenic exceeded unity under the maximum case (11 and 3.7, respectively). Only the NOAEL-based HQ exceeded unity under the average case (HQ = 2.1).

Lead

NOAEL- and LOAEL-based HQs generated for lead exceeded unity under the maximum case (18 and 1.8, respectively). Only the NOAEL-based HQ exceeded unity under the average case (HQ = 3.3).

BHC Isomers – Sum

HQs generated for BHC isomers were below unity for all scenarios.

Chlordane Isomers – Sum

NOAEL- and LOAEL-based HQs generated for chlordane exceeded unity under the maximum case (5.7 and 1.1 for, respectively). Average case HQs did not exceed unity.

DDT and Metabolites – Sum

Only the NOAEL-based HQ generated for DDT and metabolites – sum exceeded unity under the maximum case with an HQ of 2.6.

Dieldrin

HQs generated for dieldrin exceeded unity for NOAEL-based scenarios, with HQs ranging from 3.0 to 6.5 (average case and maximum case, respectively).

Endosulfans – Sum

HQs generated for endosulfans – sum were below unity for all scenarios.

Endrins – Sum, Endrin Aldehyde, and Endrin Ketone

Only the NOAEL-based HQ generated for endrins – sum exceeded unity under the maximum case with an HQ of 4.7.

Heptachlors – Sum

HQs generated for heptachlors – sum exceeded unity for NOAEL-based scenarios, with HQs ranging from 2.2 to 4.7 (average case and maximum case, respectively).

Toxaphene Parlars – Sum

HQs generated for toxaphene parlars – sum were below unity for all scenarios.

Toxaphene

HQs generated for toxaphene exceeded unity for NOAEL-based scenarios, with HQs ranging from 2.0 to 4.6 (average case and maximum case, respectively).

8.3.4.1 Summary of Risks to Insectivorous, Omnivorous, and Carnivorous Birds

NOAEL-based HQs exceeded unity for arsenic, lead, chlordane isomers – sum, DDT and metabolites – sum, dieldrin, endrins – sum, heptachlors – sum, and toxaphene under the maximum exposure scenario. LOAEL-based HQs exceeded unity for only arsenic, lead, and chlordane isomers – sum under the maximum exposure scenario.

NOAEL-based HQs exceeded unity for arsenic, lead, dieldrin, heptachlors – sum, and toxaphene under the average scenario. LOAEL-based HQs did not exceed unity under the average scenario.

8.3.5 Assessment Endpoint No. 4 – Protection of Piscivorous Mammals

The risk question developed for this assessment endpoint is:

Are the COPC concentrations in the site sediments, surface water, and fish elevated enough to cause adverse effects to the long-term health and reproductive capacity of piscivorous mammals?

The measurement endpoints for this assessment endpoint include:

1. Comparison of the fish tissue concentrations with literature-derived benchmarks protective of mammals; and
2. Food-web model.

The lines of evidence considered for the evaluation of risks to piscivorous mammals include literature-derived benchmark comparisons, and comparison of estimated daily doses to the mink (*Mustela vison*) to NOAEL and LOAEL-based TRVs.

8.3.5.1 Comparisons of Fish Tissue Body Burdens to Mammalian Piscivore TRVs

Fish at the site were caught at locations WC-115 and WC-118, as well as from the reference locations (WC-121 and WC-123). The locations from which the fish were caught are presented in Appendix IV on Figure 7-3 and fish tissue concentration are presented in Table 8-14.

Arsenic and lead concentrations in fish were lower than the lowest food-based NOAEL benchmarks developed by Sample et al. (1996) for mink of 0.383 mg/kg wet weight and 44.9 mg/kg wet weight, respectively.

Pesticides analysis was performed on fish caught only from one station at the site (i.e., 2 locations at station WC-118) and both background stations. With the exception of DDE, which was detected in all of the site-related and background fish caught at the site, no other pesticides were detected. The DDE concentrations in the site-related fish ranged from 0.015 mg/kg wet weight (0.065 mg/kg dry weight) to 0.033 mg/kg wet weight (0.13 mg/kg dry weight). Sample et al. (1996) developed a food-based NOAEL benchmark for DDT and metabolites of 4.49 mg/kg wet weight for the mink. Site concentrations are below this value.

8.3.5.2 Comparisons of Food Web Modeled Doses to TRVs

HQs for the piscivorous mammals are presented in Tables 8-15 and 8-16 for the maximum and average scenarios, respectively. More details are discussed below.

Arsenic

Only the NOAEL-based HQ generated for arsenic exceeded unity under the maximum case with an HQ of 4.3.

Lead

Only the NOAEL-based HQ generated for lead exceeded unity under the maximum case with an HQ of 1.1.

Aldrin

HQs generated for aldrin were below unity for all scenarios.

BHC Isomers – Sum

HQs generated for BHC isomers – sum were below unity for all scenarios.

Chlordane Isomers – Sum

Only the NOAEL-based HQ generated for chlordane isomers – sum exceeded unity under the maximum case scenario with an HQ of 1.9.

DDT and Metabolites

HQs generated for DDT and metabolites were at or below unity for all scenarios.

Dieldrin

Only the NOAEL-based HQ generated for dieldrin exceeded unity under the maximum case with an HQ of 3.4.

Endosulfans – Sum

HQs generated for endosulfans – sum were below unity for all scenarios.

Endrins – Sum

HQs generated for endrins – sum were below unity for all scenarios.

Heptachlors – Sum

HQs generated for heptachlors – sum were below unity for all scenarios.

Methoxychlor

HQs generated for methoxychlor were below unity for all scenarios.

Toxaphene Parlars – Sum

HQs generated for toxaphene parlars – sum were below unity for all scenarios.

Toxaphene

HQs generated for toxaphene were below unity for all scenarios.

8.3.5.3 Summary of Risks to Piscivorous Mammals

Fish concentrations of arsenic, lead, and DDE are lower than tissue benchmark values for the protection of mink. However, based on the food chain modeling results, only arsenic, lead, chlordane isomers – sum, and dieldrin would have the potential to cause adverse ecological effects under the NOAEL-based maximum case scenario. None of the COPCs show the potential for risks under the LOAEL-based maximum-case scenario or under the average-case scenario.

8.3.6 Assessment Endpoint No. 5 – Protection of Piscivorous Birds

The risk question developed for this assessment endpoint is:

Are the COPC concentrations in the site sediments, surface water, and fish elevated enough to cause adverse effects to the long-term health and reproductive capacity of piscivorous birds?

The measurement endpoints for this assessment endpoint include:

1. Comparison of the fish tissue concentrations with literature-derived benchmarks protective of mammals; and
2. Food-web model.

The lines of evidence considered for the evaluation of risks to piscivorous birds include literature-derived benchmark comparisons, and comparison of estimated daily doses to the Green heron (*Butorides virescens*) to NOAEL and LOAEL-based TRVs.

8.3.6.1 Comparisons of Fish Tissue Body Burdens to Avian Piscivore TRVs

Arsenic and lead concentrations in fish (Table 8-14) were lower than the lowest food-based NOAEL benchmarks developed by Sample et al. (1996) for piscivores (based on the belted kingfisher) of 10.1 mg/kg wet weight and 2.23 mg/kg wet weight, respectively.

The DDE fish tissue concentrations were greater than a benchmark of 0.014 mg/kg wet weight, which is the Canadian tissue residue guideline for the protection of wildlife consumers of aquatic biota value for DDT and metabolites based on exposure in the storm petrel (CCME, 2007) and the food-based NOAEL benchmark developed by Sample et al. (1996) for the belted kingfisher. This indicates the DDE concentrations in the site and background fish may pose unacceptable risks to the avian piscivores at the site and vicinity.

8.3.6.2 Comparisons of Food Web Modeled Doses to TRVs

HQs for the piscivorous birds are presented in Tables 8-17 and 8-18 for the maximum and average scenarios, respectively. Only the NOAEL-based HQ generated for lead exceeded unity under the maximum case with an HQ of 1.1. No other COPEC had HQs greater than unity.

8.3.6.3 Summary of Risks to Piscivorous Birds

Fish concentrations of arsenic and lead are lower than tissue benchmark values for the protection of piscivorous birds; however, DDE concentrations exceed the CCME and Sample et al. (1996) benchmarks for piscivorous birds. Likewise, food-web modeling did not indicate any adverse effects to piscivorous birds other than that noted for lead based on the maximum-case food chain modeling results and the NOAEL-based TRV.

8.3.7 Assessment Endpoint No. 6 – Protection of Benthic Macroinvertebrate Communities

This assessment endpoint serves to protect benthic macroinvertebrate communities at the site and vicinity to ensure that contact with and incidental ingestion of the site sediments do not negatively impact their survival and growth.

The risk questions developed for this assessment endpoint include the following:

1. Are the COPC concentrations in the site sediments/surface soils elevated in comparison to sediment benchmark values?
2. Are the COPC concentrations in the site sediments/surface soils high enough to cause mortality and/or reduction in growth of benthic macroinvertebrates in solid-phase laboratory toxicity tests?

The measurement endpoints for this assessment endpoint include:

1. Comparisons with sediment benchmark values; and
2. Aquatic invertebrate solid-phase sediment toxicity tests using the freshwater amphipod, *Hyalella azteca*.

8.3.7.1 Comparison of Sediment COPC Concentrations to Sediment Benchmark Values

Sediment concentrations were compared with sediment benchmark values in order to calculate HQs as a measurement endpoint for Assessment Endpoint No. 6 - Protection of Benthic Macroinvertebrate Communities. In this section, site-specific COPC concentrations from each sampling location were compared with sediment benchmark values in order to determine if specific locations of the site pose risks to benthic macroinvertebrates and to detect any spatial distribution patterns. A summary of the samples

that exceed the sediment ATVs is presented in Table 8-19 with the individual sample HQs presented in Appendix IV – Appendix B (Table B-2). Note that, as with the soil comparisons, the frequency of detections in Table 8-19 may be different than those presented in the summary tables (Tables 8-3 and 8-4) as the individual sample HQ calculations treated primary and field duplicate results separately; whereas, the frequency of detection in the summary tables treated primary and field duplicates as one sample.

Arsenic

The ATV selected for arsenic was the State of Florida probable effect concentration (PEC) of 33 mg/kg dry weight (FDEP, 2003). The selected ATV was more conservative than the State of Washington Sediment Quality Standard of 57 mg/kg (Washington State, 2008). The PEC was calculated based on aquatic toxicity tests and is a concentration above which adverse ecological effects are likely to occur. The maximum arsenic concentration in the site sediments of 1,500 mg/kg generated an HQ of 45.5. Arsenic concentrations in 119 out of 217 sediment samples had concentrations that exceeded the ATV. Background concentrations generated HQs less than unity.

Lead

The ATV selected for lead was the State of Florida PEC of 130 mg/kg dry weight (FDEP, 2003). The maximum lead concentration in sediment of 1,600 mg/kg generated an HQ of 12.3. Nineteen of 62 samples generated HQs greater than unity. Of these, only one sample had an HQ greater than 10. Background concentrations generated HQs less than unity.

Aldrin

The ATV for aldrin was the Ontario lowest effect level of 0.002 mg/kg dry weight (Persaud et al. 1992). The one detected concentration of 0.011 mg/kg generated an HQ of 5.5. Aldrin was not detected in background.

Benzene Hexachloride (BHC) Isomers

Various isomers of benzene hexachloride (i.e., alpha, beta, delta, and gamma) were detected in the sediments at the site. For the purposes of this risk characterization, the concentrations of each isomer were compared to the FDEP PEC ATV of 0.005 mg/kg dry weight for gamma-BHC (FDEP, 2003). The isomers were not summed as the gamma isomer is most toxic and assuming that each of the other isomers contributes the same toxicity is overly conservative.

All but the delta isomer were detected in approximately 40 percent of the sediment samples; whereas, delta-BHC was only detected in approximately 12 percent of samples. BHC isomers exceeded the benchmark as follows:

- alpha-BHC: 14/21 samples – maximum HQ = 30, the rest of the HQs were below 10;
- beta-BHC: does not exceed the benchmark;
- delta-BHC: 6/8 samples – maximum HQ = 5.8; and
- gamma-BHC: 14/21 samples – maximum HQ = 28, all but two HQs were below 10.

BHC isomers were not detected in background.

Chlordane

The chlordane isomers (alpha and gamma) were summed and compared with an FDEP PEC ATV of 0.018 mg/kg dry weight (FDEP, 2003). The calculated total chlordane concentration in sediment of 28 mg/kg generated an HQ of 1,560. Thirty-one of 37 samples exceeded the ATV. Of these, 23 HQs were above 10, with 14 of those above 100. Background concentrations generated HQs less than unity.

DDT and Metabolites

The ATV for DDT and metabolites was the state of Florida PEC of 0.570 mg/kg dry weight (FDEP, 2003). Twenty-one out of 52 samples generated HQs greater than unity. Of these, 5 HQs were above 10, 1 of which was above 100. Background concentrations generated HQs less than unity.

Dieldrin

The ATV for dieldrin was the State of Florida PEC of 0.062 mg/kg dry weight (FDEP, 2003). The maximum detected dieldrin concentration of 9.1 mg/kg generated an HQ of 147. Twenty-eight of 40 samples exceeded the ATV. Of these, 11 HQs were above 10, with 1 above 100. Background concentrations generated HQs less than unity.

Endosulfan I

Endosulfan I was compared with a USEPA SEC ATV of 0.0029 mg/kg dry weight for endosulfan (USEPA, 1996). Endosulfan I was detected in only 3 of 57 samples. All three of these samples had concentrations that exceeded the ATV, with a maximum HQ of 72.4. The other two HQs were less than 10. Endosulfan I was not detected in background.

Endosulfan II

Endosulfan II was compared with a USEPA SEC ATV of 0.0029 mg/kg dry weight for endosulfan (USEPA, 1996). Endosulfan II was detected in 27 of 56 samples. Twenty-seven of 29 concentrations exceeded the ATV, with a maximum HQ of 931. Only two HQs greater than unity were less than 10. Fourteen were between 10 and 100 and 11 were greater than 100. Endosulfan II was not detected in background.

Endosulfan Sulfate

Endosulfan sulfate was compared with a USEPA Region 5 ESL ATV of 0.0346 mg/kg dry weight for endosulfan sulfate (USEPA, 2003). Eighteen of 21 concentrations exceeded the ATV, with a maximum HQ of 15.9. Of these, only two HQs were greater than 10. Endosulfan sulfate was not detected in background.

Endrin, Endrin Aldehyde, and Endrin Ketone

Endrin, endrin aldehyde, and endrin ketone concentrations were compared with an FDEP ATV of 0.210 mg/kg dry weight for endrin (FDEP, 2003). These compounds were not summed as their relative toxicity is not well known. Endrin-related compounds were detected in only 13-31 percent of samples. Endrin-related compounds exceeded the benchmark as follows:

- Endrin: 4/10 samples – maximum HQ = 27.1, the rest of the HQs were below 10;
- Endrin aldehyde: 6/7 samples – maximum HQ = 6.19; and
- Endrin ketone: 2/18 samples – maximum HQ = 36.2, the other HQ was below 10.

Endrin was not detected in background.

Heptachlor

Heptachlor concentrations were compared with an FDEP ATV of 0.016 mg/kg dry weight for heptachlor epoxide (FDEP, 2003). Five of 12 samples generated HQs greater than unity, with a maximum HQ of 31.9. The rest of the HQs were less than 10. Heptachlor was not detected in background.

Heptachlor Epoxide

Heptachlor epoxide concentrations were compared with an FDEP ATV of 0.016 mg/kg dry weight (FDEP, 2003). Eighteen of 21 samples generated HQs greater than unity, with a maximum HQ of 17.5. Four of these samples generated HQs greater than 10. Heptachlor epoxide was not detected in background.

Methoxychlor

Methoxychlor concentrations were compared with an USEPA Region 5 ESL ATV of 0.0136 mg/kg dry weight (USEPA, 2003). Methoxychlor was detected in only 4 of 44 samples, with 2 concentrations out of 4 exceeding the ATV. The maximum HQ was 44.9, with the other HQ being less than 10. Methoxychlor was not detected in background.

Toxaphene

Toxaphene concentrations were compared with an USEPA ATV of 0.028 mg/kg dry weight (USEPA, 1996). Toxaphene exceeded the ATV in all 28 of the samples in which it was detected. The maximum HQ was 6,070. Twenty-six of the samples had HQs greater than 10; 16 of which were greater than 100.

Toxaphene parlars – sum concentrations also were compared with the USEPA ATV. Ten of 14 samples had HQs greater than 1, with the maximum HQ being 239. Eight of the samples had HQs greater than 10, 1 of which was greater than 100. Toxaphene was not detected in background.

8.3.7.2 Sediment Toxicity Data – Results

Whole sediment toxicity tests were used as a measurement endpoint to assess the risk to benthic macroinvertebrates. The toxicity tests were performed by USEPA Region 4 ESAT toxicologists using the freshwater amphipod, *Hyalella azteca* (in 2007 and 2010) and the oligochaete, *Lumbriculus variegatus* (in 2010). The toxicity tests were conducted following USEPA guidelines described in *Methods for Measuring the Toxicity and Bioaccumulation of Sediment-associated Contaminants with Freshwater Invertebrates* (USEPA, 2000).

The selection of sediment sample locations for both the 2007 and 2010 toxicity tests were based on known arsenic concentrations. Every effort was made to ensure a wide range of arsenic concentrations were present in these samples. The toxicity tests for *H. azteca* in 2007 were performed with five site sediments (WC-105-SD, WC-106-SD, WC-109-SD, WC-115-SD, and WC-118-SD), two field reference or background sediments (WC-121-SD and WC-123-SD), and a laboratory control sediment (control). The 2010 toxicity tests for *H. azteca* were performed with four site sediments (WC-160, WC-174, WC-175, and WC-177), one field reference (WC-125), and one laboratory control sediment.

The sediment used as the control for the toxicity tests was obtained from the Ogeechee River, Georgia. A full scan analysis was conducted to verify that the sediment was appropriate for use as a control in the toxicity tests. The control sediment did not contain any chemicals of interest at the site. The TOC content of the control sediment was 8.9 percent.

Acute toxicity tests for *L. variegatus* were performed in 2010 with two site sediments (i.e., WC-175 and WC-216) and one laboratory control sediment (the sample control sediment that was used for the 2010 *H. azteca* tests). A 28-day bioaccumulation test was also conducted in 2010 using *L. variegatus* and sediments collected at 4 locations (WC-125-ref, WC-160, WC-175 and WC-177); the results of this study are used to assess exposure for piscivorous mammals and birds and are not evaluated as part of this assessment endpoint.

The sediment toxicity test reports containing information on test dates, water quality measurements, test observations, specific test methods, and any modifications are provided in Appendices C and D.

2007 Test

The test endpoints for the acute toxicity tests with *H. azteca* were survival and growth (optional). The results of the *H. azteca* sediment toxicity tests are presented in Table 8-20. Statistically significant differences ($P=0.05$) were noted in the survival of *H. azteca* in the sediment sample from station WC-105-

SD when compared to the laboratory control and field reference average. WC-105-SD had the lowest TOC concentration (0.36 percent) of all the site-related samples, the other four site-related samples had TOC concentration ranging from 2 to 8.5 percent. The low TOC concentration associated with WC-105-SD may have contributed to higher contaminant bioavailability and subsequent toxicity. In terms of growth, statistically significant differences ($P=0.05$) were noted in the growth of *H. azteca* in the sediments from station WC-106-SS and WC-109-SS when compared to the reference station average or control. It must be noted that these two samples were sediment but collected as soil because the creek bed was dry at the time of sampling.

2010 Tests

Hyalella azteca

The test endpoints for the acute toxicity tests with *H. azteca* were survival and growth and were the same as in the 2007 test. The results of the *H. azteca* sediment toxicity tests are presented in Table 8-21. Statistically significant differences ($P=0.05$) were noted in the survival of *H. azteca* in the sediment samples from stations WC-160 and WC-175 when compared to the laboratory control and field reference average; therefore, growth was not calculated for these samples. For the other samples, statistically significant differences ($P=0.05$) in growth were not observed. It should be noted that pH levels for site samples and the reference samples were at or below 4, which is below the level recommended for this test, but pH did not appear to have any effects on the test results. TOC concentrations were fairly uniform in the site-related samples ranging from 2.2 to 3.4 percent, while grain size distribution was variable across samples with no discernible relation with observed toxicity.

Lumbriculus variegatus

The 96-hour toxicity tests used a sample with a high arsenic concentration (WC-216) and one with a mid-level arsenic concentration (WC-175). The WC-216 sample was found to be acutely toxic and resulted in 100 percent mortality. Survival was significantly less ($P=0.05$) than the survival in the laboratory control sediment. Sample WC-216 was originally collected to perform dilution series tests but since a good concentration gradient was achieved with test samples, the dilution series tests were abandoned. Refer to Table 8-27 for specific details.

8.3.7.3 Sediment Toxicity Tests – Discussion

Toxicity test results of mixed contaminant samples are sometimes difficult to interpret because different chemicals, with different physico-chemical parameters and/or characteristics come into play. The results of the *H. azteca* sediment toxicity tests and estimates of associated NOECs and LOECs from the tests are discussed in the following sections. The rationale for selecting the NOEC and LOEC is also provided, if necessary. NOECs and LOECs are typically developed in association with toxicity tests that are looking

at effects association with exposure to one chemical at a time. The use of this terminology in conjunction with multiple chemical exposure simultaneously needs to be viewed with caution.

The direct toxicity of the pesticides to benthic invertebrates was not the focus when the sediment toxicity tests were designed (see previous discussion on sample location selection criteria). However, in the 2010 test, high pesticide levels were associated with samples that also had high arsenic levels; this allowed for the development of several NOECs and LOECs. Because the direct toxicity of the pesticides was not a subject of this investigation, these results should be used cautiously.

In 2010, the bioaccumulation of pesticides in the benthic community were assessed as part of the *L. variegatus* toxicity and bioaccumulation study. The following discussion of the 2007 and 2010 *H. azteca* toxicity tests, specifically evaluates the potential impacts of arsenic, lead, DDT, and toxaphene on the OU5 benthic community.

Arsenic

2007

A review of the chemical analysis results indicated that a true concentration gradient was obtained for the arsenic in the sediments used in the toxicity tests. Arsenic concentrations in the site sediments ranged from 6.2 mg/kg in sample WC-118-SD to 280 mg/kg in sample WC-106-SS. The field reference sediment concentrations were 1.1 U mg/kg and 0.69 mg/kg. In terms of survival, the sample that showed a significant reduction in survival (i.e., sample WC-105-SD) had an arsenic concentration of 6.8 mg/kg but the sample with the highest arsenic concentration (i.e., WC-106-SS with a concentration of 280 mg/kg) did not show any acute effects on survivorship. This indicates that the significant differences in mortality observed in sample WC-105-SS was not due to arsenic but to some other COPC or environmental factor(s).

In terms of growth, the lowest arsenic concentration that caused a reduction in growth was 54 mg/kg at station WC-109-SS and the highest concentration which did not cause a reduction in growth was reference sample WC-115-SD at a concentration of 14 mg/kg. Based on the above synopsis, the NOEC for growth is >14 mg/kg and the LOEC for growth is 54 mg/kg.

2010

A true concentration gradient was obtained for arsenic in sediments used in the 2010 toxicity test. Arsenic concentrations in site sediments ranged from 36 mg/kg in samples WC-174 to 610 mg/kg in sample WC-175. The field reference sediment concentration was 5 mg/kg. In terms of survival, two samples (WC-160 and WC-175) had survival significantly lower ($P=0.05$) than the reference and laboratory control. The arsenic concentrations for these two samples were the highest observed in the test (330 mg/kg and 610 mg/kg, respectively). The highest arsenic concentration observed in a sample

with no significant decrease in survival was 200 mg/kg, recorded for sample WC-177. Based on these results, the NOEC for survival was >200 mg/kg and the LOEC for survival was <330 mg/kg.

Due to low survival in samples WC-160 and WC-175, growth was only assessed for two site samples (WC-174 and WC-177), the reference sample and the laboratory control. There were no significant differences observed in growth between the two site samples and the reference and control samples.

Lead

2007

The chemical analysis results indicated that a concentration gradient was obtained for the lead in the sediments used in the toxicity tests. Concentrations of lead used in the sediment toxicity tests ranged from 5.8 mg/kg in sample WC-105-SD to 450 mg/kg in sample WC-106-SS. In terms of survival, the sample which showed a significant reduction in survival (i.e., sample WC-105-SD) had the lowest lead concentration; but the sample with the highest lead concentration (i.e., WC-106-SS) did not show any effects on survivorship. This indicates that the significant differences in mortality observed in sample WC-105-SS was not due to lead but to some other analyte(s) or environmental factor(s).

In terms of growth, the lowest lead concentration that caused a reduction in growth was 41 mg/kg at station WC-109-SS and the highest concentration that did not cause a reduction in growth was reference sample WC-118-SD at a concentration of 19 mg/kg. Based on the above synopsis, the NOEC for growth is >19 mg/kg and the LOEC for growth is 41 mg/kg.

2010

A true concentration gradient was obtained for lead in sediments used in the 2010 toxicity test. Lead concentrations in site sediments ranged from 43 mg/kg in samples WC-174 to 530 mg/kg in sample WC-175. The field reference sediment concentration was 6.8 mg/kg. In terms of survival, two samples (WC-160 and WC-175) had survival significantly lower ($P=0.05$) than the reference and laboratory control; the lead concentrations for these two samples were the highest observed in the test (380 mg/kg and 530 mg/kg, respectively). The highest lead concentration observed in a sample with no significant decrease in survival was 260 mg/kg recorded for sample WC-177. Based on these results, the NOEC for survival was > 260 mg/kg and the LOEC for survival was < 380 mg/kg.

Due to low survival in samples WC-160 and WC-175, growth was only assessed for two site samples (WC-174 and WC-177), the reference sample and the laboratory control. There were no significant differences observed in growth between the two site samples and the reference and control samples.

DDTr

2007

As stated earlier, the direct toxicity of DDTr was not a subject of test. However, based on the toxicity results, the adverse effects in the benthic invertebrates for DDTr follow the same pattern as the metals which were assessed for direct toxicity. In terms of survivorship, the samples with the highest DDTr concentrations of 4,870 µg/kg (i.e., WC-106-SS) did not show any adverse effects on survival. The sample with the next highest concentration of DDTr of 132 µg/kg (i.e., WC-105-SD) showed a significant reduction in survival and the sample with the third highest concentration of 59.5 µg/kg (i.e., WC-109-SS) did not show any significant reduction in growth. The TOC content in sample WC-105-SD was 0.72 percent and the TOC in samples WC-106-SS and WC-109-SS were 1.9 and 4.5 percent, respectively. The low TOC contents in the latter two samples do not account for the toxicity observed in sample WC-105-SD. Based on the above synopsis, it can be inferred that the reduction in survival observed in the sample from station WC-105-SD was not due to DDTr, but to some other analyte(s) or environmental factor(s).

The lowest DDTr concentrations that caused a reduction growth was 59.4 µg/kg (station WC-109-SS) and the highest concentration that did not cause a reduction in growth was sample WC-115-SD at a concentration of 29.2 µg/kg. Based on the above results, the NOEC for DDTr for growth is >29.2 µg/kg and the LOEC for growth is <59.4 µg/kg.

2010

A true concentration gradient was obtained for DDTr in sediments used in the 2010 toxicity test. DDTr concentrations in site sediments ranged from 103 µg/kg in samples WC-174 to 4,465 µg/kg in sample WC-160. DDTr were not detected in the field reference sediment. In terms of survival, two samples (WC-160 and WC-175) had survival significantly lower ($P=0.05$) than the reference and laboratory control; the concentrations of DDTr for these two samples were the highest observed in the test (4,465 µg/kg and 2,067 µg/kg, respectively). The highest concentration of DDTr observed in a sample with no significant decrease in survival was 1,030 µg/kg recorded for sample WC-177. Based on these results, the NOEC for survival was > 1,030 µg/kg and the LOEC for survival was < 2,067 µg/kg.

Due to low survival in samples WC-160 and WC-175, growth was only assessed for two site samples (WC-174 and WC-177), the reference sample and the laboratory control. There were no significant differences observed in growth between the two site samples and the reference and control samples.

Toxaphene

2007

Due to analytical difficulties, toxaphene results for the 2007 toxicity tests were not usable.

2010

A true concentration gradient was obtained for toxaphene in sediments used in the 2010 toxicity test. Toxaphene concentrations in site sediments ranged from 760 µg/kg in samples WC-174 to 17,000 µg/kg in sample WC-160. Toxaphene not detected in the field reference sediment. In terms of survival, two samples (WC-160 and WC-175) had survival significantly lower ($P=0.05$) than the reference and laboratory control; the toxaphene concentrations for these two samples were the highest observed in the test (17,000 µg/kg and 8,700 µg/kg, respectively). The highest toxaphene concentration observed in a sample with no significant decrease in survival was 6,700 µg/kg recorded for sample WC-177. Based on these results, the NOEC for survival was $> 6,700$ µg/kg and the LOEC for survival was $< 8,700$ µg/kg.

Due to low survival in samples WC-160 and WC-175, growth was only assessed for two site samples (WC-174 and WC-177), the reference sample and the laboratory control. There were no significant difference observed in growth between these two site samples and the reference and control samples.

Other Pesticides

The toxicity results for the remaining pesticides basically mirrored those of arsenic, lead, DDT_r, and toxaphene and will not be discussed in detail in this section.

8.3.7.4 Summary of Risks to Benthic Macroinvertebrates

Based on the HQs calculated from the sediment concentrations and the sediment benchmark values, 80 locations (represented by 119 samples) had HQs that were greater than unity for arsenic and may have sediment concentrations high enough to cause adverse effects to sediment macroinvertebrates (see Appendix IV – Appendix B [Table B-2]). In addition to the 80 locations with elevated arsenic levels, 26 locations had lead and/or pesticide concentrations with HQs exceeding unity (see Table 8-23). Of these locations, three did not have elevated arsenic levels (i.e., WC-105, WC-166, and WC-167).

The results of the solid-phase sediment toxicity tests with *H. azteca* indicate that the levels of COPCs in the site sediments were high enough to cause adverse effects to the benthic macroinvertebrate community. Because survival is considered a more significant ecological effect than growth, the following list of survival-based NOECs and LOECs are provided.

- Arsenic – NOEC for survival was estimated to be 200 mg/kg and the LOEC for survival was estimated to be 330 mg/kg.

- Lead – NOEC for survival was estimated to be 260 mg/kg and the LOEC for survival was estimated to be 380 mg/kg.
- DDT_r – NOEC for survival was estimated to be 1,030 µg/kg and the LOEC for survival was estimated to be 2,067 µg/kg.
- Toxaphene – NOEC for survival was estimated to be 6,700 µg/kg and the LOEC for survival was estimated to be 8,700 µg/kg.

8.3.8 Assessment Endpoint No. 7 – Protection of Fish Populations

This assessment endpoint provides for the protection of fish communities to ensure that fish populations inhabiting the wetlands and creeks at the site and vicinity are not adversely affected by contaminants found in the surface waters and sediments.

The risk questions for this assessment endpoint include:

1. Are the COPC concentrations in surface water greater than the National Recommended Water Quality Criteria (NRWQC) or other freshwater surface water benchmarks?;and
2. Are concentrations of COPCs in the fish tissues at the site and vicinity elevated enough to cause adverse effects to fish?

The measurement endpoints for this assessment endpoint include:

1. Comparison of surface water chemical concentrations with benchmark values;
2. Chronic toxicity tests with surface water from the site and vicinity using the aquatic invertebrate, *Ceriodaphnia dubia* as a surrogate for the fish community; and
3. Comparison of the fish tissue concentrations with literature-derived TRVs.

8.3.8.1 Risk Evaluation Using Benchmark Values

For this risk characterization, seven surface water samples (i.e., four site waters, one duplicate, and two background samples) were used (Table 8-24). The NRWQC (USEPA, 2009b) were used as the initial benchmark values comparison to the COPCs in surface water. Benchmarks from other sources were used if NRWQC benchmarks were not available.

Metals

Arsenic was detected in only two of the surface water samples. Both detections generated HQs less than unity when compared with the NRWQC of 150 µg/L. Lead was detected in only one surface water sample at a concentration of 3.8 µg/L. The detected lead concentration exceeded the NRWQC of 2.5 µg/L. The detection limit for lead in all of the surface water samples was higher than the NRWQC; however, it should be noted that total water concentrations were compared to dissolved benchmarks.

Pesticides

The organochlorine pesticides were detected in only one of the surface water samples (i.e., WC-105-SW). Pentachlorophenol, which was not a COPC, was also detected in this sample. Other than for DDD, beta-BHC, and gamma-BHC, the pesticide results in this sample were either rejected (R-qualified as in the case of DDT) or NJ qualified. Detected, NJ, or SQL concentrations of DDD, DDE, DDT, gamma-BHC, and endrin ketone were above the ATV. The NJ-qualified concentration of endosulfan II was below the ATV, but the SQLs for the remaining samples were above the ATVs. The remaining pesticides (i.e., beta-BHC, delta-BHC, endosulfan I, and endosulfan sulfate) had concentrations (J-, NJ-, and/or U-qualified) below the ATVs. This indicates that these four pesticides alone are unlikely to cause adverse effects to the ecological receptors in the surface waters at the site.

pH

The pH of the surface water samples used in the toxicity tests was measured in the toxicity laboratory. With the exception of one sample (i.e., WC-105-SW with a pH of 6.79) all of the surface water samples including the background samples were less than the minimum NRWQC of 6-9. Low pH has the potential to mobilize certain metals in solution and make them more bioavailable.

8.3.8.2 Risk Evaluation Using Site-Specific Surface Water Toxicity Data

Chronic surface water toxicity tests were used as a measurement endpoint to assess the risk to the fish populations at the site. The toxicity tests were performed by USEPA Region 4 ESAT toxicologists using the freshwater daphnid, *Ceriodaphnia dubia*. The toxicity tests were conducted following USEPA guidance described in *Short-Term Methods for Estimating the Chronic Toxicity of Effluents and Receiving Waters to Freshwater Organisms* (USEPA, 1994). The duration of the chronic toxicity test was 7 days or until 60 percent of the control females had produced a minimum of three broods. The test endpoints for the *C. dubia* chronic toxicity tests were survival and reproduction.

The toxicity tests were performed with three surface water samples (i.e., WC-105-SW, WC-115-SW, and WC-118-SW), two field reference or background surface water samples (WC-121-SW and WC-123-SW), and a laboratory control sample (control). The control water used in the surface water toxicity tests was dilute mineral water comprising 20 percent Perrier® water and 80 percent Milli-Q water (i.e., deionized

water that had been further purified by passing through a Milli-Q deionizing system) mixture with pH, hardness, alkalinity and conductivity values of approximately 7.74 standard units (s.u.), 80 milligrams per liter (mg/L) as calcite (CaCO_3), 65 milligrams per liter (mg/L) as CaCO_3 , and 182 micromhos per centimeter, respectively.

The locations from which the surface water toxicity test samples were collected are presented in Appendix IV on Figure 7-3. The pH of the samples was measured prior to use in testing. A pH of 6 is the minimum pH required for *C. dubia* survivorship during a toxicity test. The pH WC-115-SW had a pH of 3.69; therefore, this sample was separated into two portions and one portion had the pH adjusted to 7 and was run as a separate sample. The toxicity test report containing information on test dates, water quality measurements, test observations, specific test methods, and any modifications is presented in the *Field Investigation Data Report* (USEPA, 2008a). The results of the *C. dubia* toxicity tests are presented in this report in Table 8-25.

For the purpose of this risk characterization, results compared to the field reference samples will be determinative, as opposed to those compared with controls. Statistically significant differences ($P=0.05$) were noted in the survival of *C. dubia* in the surface water samples from stations WC-105-SW and WC-115-SW (native sample with a pH of 3.69 s.u.) when compared to the laboratory control and field reference average. In terms of reproduction, statistically significant differences ($P=0.05$) were noted in the reproduction of *C. dubia* in the sample from station WC-115-SW (pH adjusted to 7 s.u.) when compared with the reference station average and the control. Also, reproduction in the sample from station WC-118-SW was significantly less ($P=0.05$) than reproduction in the control sample but not than the reference average.

Arsenic was detected in both of the samples where toxicity was observed (i.e., WC-105-SW at a concentration of 72 $\mu\text{g/L}$ and WC-115-SW at a concentration of 6 $\mu\text{g/L}$) in the site-specific toxicity tests. However, the concentrations of arsenic in these two samples were below the chronic NRWQC. Therefore, arsenic alone may not have been responsible for the observed adverse effects. Lead was detected in only one of the two samples that caused toxicity to *C. dubia* (i.e., WC-115-SW at a concentration of 3.8 micrograms per liter [$\mu\text{g/L}$]); however, the SQL for lead in WC-105-SW was higher than the ATV. Other confounding issues with the potential attribution of effects is that total metals concentrations were compared with dissolved criteria and that the lead criterion is hardness-based and a default hardness value (100 mg CaCO_3/L) was assumed as site-specific hardness data were not available.

The other toxic sample (WC-105-SW) was the only sample which had any detection of pesticides. However, with the exception of beta- and delta-BHC, endosulfan I, and endosulfan sulfate, the SQLs for pesticides were higher than the ATVs in the remaining samples.

Lastly, another confounding factor in the interpretation of results is that the pH tended toward acidity in most samples, which would increase the bioavailability of metals; but, with the exception of sample WC-105-SW, based on the NRWQC, pH in and of itself was less than the optimal required for the survival of aquatic organisms.

8.3.8.3 Comparison of Fish Body Burdens to TRVs

In aquatic environments, most of the organic and metal chemicals may be bound to sediment and not available for uptake by fish. This appears to be the case in this study in which very few organic chemicals were detected in the fish tissues (Table 8-14). The detected arsenic, lead, and DDE concentrations in the fish tissues were compared with TRVs, if available. It must be noted that fish were not caught at the stations with the highest COPC concentrations because the creeks were dry. If the areas with the highest COPC concentrations are wet enough during parts of the year to support a fish community, risks noted below may be underestimated.

Arsenic

The concentrations of arsenic in the site fish ranged from 0.054 mg/kg wet weight to 0.3 mg/kg wet weight. Whole fish body residues of arsenic from fish diets have been reported to be in the range of 0.4 mg/kg wet weight to 21.6 mg/kg wet weight with no adverse effects (Nichols et al., 1984; Oladimeji et al., 1984; Cockell and Hilton, 1988).

Lead

The detected concentrations of lead in the site fish ranged from 0.11 mg/kg wet weight to 0.3 mg/kg wet weight (or 1.2 mg/kg dry weight). The fish tissue concentrations were less than those reported in the literature, which had no effects on fish species (Maddock and Taylor, 1980).

Pesticides

As noted previously, pesticides analysis was performed on fish caught only from one station at the site (i.e., two locations at station WC-118) and both background stations. DDE was the only pesticide detected in fish. The DDE concentrations in the site-related fish ranged from 0.015 mg/kg wet weight (0.065 mg/kg dry weight) to 0.033 mg/kg wet weight (0.13 mg/kg dry weight). The background-related fish tissue DDE concentrations ranged from 0.028 mg/kg wet weight (0.12 mg/kg dry weight) to 0.061 mg/kg wet weight (0.23 mg/kg dry weight). Behavioral and reproductive effects seem to be noted beginning at 40 mg DDE/kg wet weight adult whole body fish residue (Johnson et al., 2007 and Peterson, 1973).

8.3.8.4 Summary of Risks to the Fish Community

The results from two (i.e., comparisons to NRWQCs and *C. dubia* toxicity test results) of the three lines of evidence indicate that there may be unacceptable risks to the fish community at the site.

Comparisons with NRWQC indicate that concentrations of lead and pesticides in the surface water may cause adverse effects in the aquatic community. However, these results are confounded by the lack of site-specific dissolved metals concentrations and hardness data, the small number of pesticide detects coupled with detection limits higher than ATVs, and pH values lower than the NRWQC recommended values. *C. dubia* mortality was 100 percent in samples WC-105-SW and WC-115-SW (native sample), and reproduction was depressed in WC-115-SW (pH adjusted sample) versus the site-specific reference samples. The pH in the native WC-115-SW samples was too low to expect survivorship. Although qualified, the concentrations of pesticides and lead may have contributed to the toxicity seen in sample WC-105-SW. Lead (but not arsenic) was detected in WC-115-SW at a concentration above the ATV; and as noted above, pesticides were not detected, but detection limits were elevated. Therefore, the reproductive effects observed in WC-115-SW (pH adjusted) may be from lead, pesticides, or some unknown stressor. Residue concentrations in fish do not appear to be high enough for toxicity to be exhibited. Overall, some adverse effects to the aquatic community may be expected; but the extent and causation are unclear.

8.3.9 Assessment Endpoint No. 8 – Protection of Reptiles and Amphibians

This assessment endpoint was not evaluated individually because available toxicity values for reptiles and amphibians are limited. It is assumed that surface water comparisons with aquatic community benchmarks would be reasonable surrogate endpoints for reptiles and amphibians at this site; therefore, herptiles are accounted for under Assessment Endpoint No. 7.

8.4 RISK DESCRIPTION

One of the key outputs of the risk characterization step are contaminant concentrations in each environmental medium that bound the threshold for estimated adverse ecological effects, given the uncertainty in the data and the models used (USEPA, 1997a). These lower- and upper-bound thresholds are the RGOs. Information relating RGOs with the potential ecological significance of the estimated risks is also discussed.

8.4.1 Remedial Goal Option Development

Remedial goal options are calculated for each chemical within an endpoint that indicated a potential risk. The calculation of RGOs based on food-web modeling and toxicity tests are discussed below.

Food Web-based RGOs

Food web-based RGOs were calculated for receptor/chemical combinations with maximum case LOAEL-based HQs greater than 1.0. For food web-based RGOs, the environmental concentration of a contaminant is calculated for an HQ equal to unity. The lower- and upper-bound thresholds were derived using the NOAEL and LOAEL TRVs used in the food-web models and bioaccumulation factors for relating tissue to soil concentrations. Equations used for calculating RGOs are presented in Table 7-26.

Bioaccumulation Factors (BAFs)

Soil-to-earthworm BAFs were needed to relate earthworm concentrations to soils concentrations. Because site-specific bioaccumulation data were available for *Eisenia*, these data were used to determine site-specific BAFs. In instances where the COPC concentrations were not detected in one or the other medium (i.e., soil or worm), site-specific BAFs could not be calculated. Otherwise, COPC-, species- and sample-specific-BAFs were calculated as follows:

$$BAF = \frac{C_{\text{tissue}} \times CF}{C_{\text{abiotic medium}}}$$

Where:

BAF	=	Bioaccumulation factor kg dw soil/kg dw tissue
C_{tissue}	=	COPC tissue concentration (mg/kg ww)
CF	=	Wet weight to dry weight conversion factor (0.8 for <i>Eisenia</i>)
$C_{\text{abiotic medium}}$	=	COPC abiotic medium concentration (mg/kg dw)

Site-specific soil invertebrate BAFs were calculated as presented in Table 8-27.

In cases where site-specific *Eisenia*-based BAFs could not be calculated, the following hierarchy was used to estimate soil concentrations:

- literature-based measured soil-to-soil invertebrate BAF values; and
- COPC-specific soil-to-soil invertebrate concentration regression equations.

BAFs and supporting input parameters for the regression equations are presented in Table 8-28.

The NOAEL-based RGOs provide estimates of the highest abiotic media concentrations that would not result in unacceptable adverse effects to ecological receptors, but they do not provide concentrations where unacceptable adverse effects might occur. On the other hand, the LOAEL-based RGO ranges provide information on the lower end of the effect concentrations that may potentially cause harm to the receptors. Therefore, providing data on both the NOAEL- and LOAEL-based RGOs provides a

reasonable estimate of the range of abiotic media concentrations that will result in no unacceptable risks to the ecological receptors at the site. The results of the surface soil threshold calculations are presented in Table 8-29.

Soil Toxicity Test RGOs

Based on the results of the 2010 14-day toxicity tests with *E. foetida*, the following NOEC-based RGOs were developed for survival of the soil invertebrate community.

- Arsenic – 1,500 mg/kg.
- Lead – 1,600 mg/kg.

Sediment Toxicity Test RGOs

Based on the results of the solid-phase sediment toxicity tests with *H. azteca*, the following RGOs were developed to protect against decreased survival in the benthic macroinvertebrate community.

- Arsenic – NOEC = 200 mg/kg and the LOEC = 330 mg/kg.
- Lead – NOEC = 260 mg/kg and the LOEC = 380 mg/kg.

Surface Water Toxicity Test RGOs

RGOs based on the *C. daphnia* test were not developed because attribution of effects was not possible due to limitations in the surface water data set (e.g., small sample numbers due to lack of habitat, data usability issues). As such surface water RGOs were not developed, but is expected that cleanup based on RGOs for soil/sediment would effectively reduce the amount of contaminant able to be partitioned to surface water.

8.4.2 Remedial Goal Option Presentation

Based on site-specific toxicity tests and food-web modeling, RGOs for soil and sediment were developed as noted above. The ranges of the RGOs for each COPC are presented in Table 8-30.

8.4.3 Additional Risk Information

In this section, the RGOs are placed in context with a description of their extent, magnitude, and potential ecological significance. In addition, a discussion of how arsenic concentrations correlate with the other COPCs is provided, as many of the sampling decisions were made based on arsenic concentrations. When applicable, included in the discussion of each medium is the areal extent of contamination above

the thresholds, degree to which the thresholds are exceeded, and the potential for natural recovery once the sources of contamination are removed.

8.4.3.1 Correlation of Arsenic Concentrations with Other COPC Concentrations

While metal and organic chemical analyses have been conducted for many of the soil and sediment samples collected in OU5, emphasis has been placed on using arsenic levels to identify the extent of site-related contamination and to provide a range of contamination for toxicity testing. This approach of using arsenic as an indicator chemical is valid if a strong relationship exists between arsenic levels in soil/sediments and other COPCs (primarily pesticides). To assess the strength of these relationships, correlation analyses were run between arsenic concentrations and co-occurring concentrations of each COPC individually. The results of this analysis are presented in Appendix IV – Appendix E. With the exception of endosulfan sulfate (a breakdown product of endosulfan) and endrin aldehyde (an impurity and breakdown product of endrin), there was a statistically significant, positive correlation observed. The strongest correlation occurred between arsenic and the following: lead, alpha-BHC, DDT and metabolites, chlordane, dieldrin, endosulfan II, and toxaphene.

8.4.3.2 Surface Soil Mediated Exposures

Endpoints that addressed the soil mediated exposures were Assessment Endpoints:

No. 1 – Protection of Soil Invertebrates;

No. 2 – Protection of Insectivorous, Omnivorous, and Carnivorous Mammals; and

No. 3 – Protection of Insectivorous, Omnivorous and Carnivorous Birds

For Assessment Endpoint No.1, potential risks were noted based on the soil concentration comparisons with ATVs. However, the results of the toxicity tests with *E. foetida* that used site soils did not find significant adverse effects on survival. Arsenic concentrations within the toxicity tests samples ranged from 54 mg/kg to 1,500 mg/kg; with the maximum being the highest arsenic concentration found in OU5 soils. Due to the lack of measurable effects to *E. foetida* at existing contaminant levels, the use of soil toxicity-based RGOs is not recommended.

However, Assessment Endpoint Nos. 2 and 3 (i.e., food-web based HQs lines of evidence) indicate the potential for risk to mammals, particularly from arsenic, chlordane isomers – sum, DDT and metabolites – sum, dieldrin, heptachlors – sum, and toxaphene in mammals as the average-case LOAEL-based HQs were greater than unity (Table 8-11). Average-case LOAEL-based HQs generated for birds were not greater than unity (Table 8-13).

However, given the conservatism incorporated into the insectivore models (e.g., assumption of an FI of 1 from the drainage when it is seasonally dry), the risks to insectivore populations are likely overestimated from exposure to COPCs in soil.

When comparing the most protective RGO calculated for either insectivorous mammals or birds (derived from NOAEL-based TRVs) to site concentrations, all but four samples (i.e., WC-101, WC-103, WC-104, and WC-117) have concentrations of at least one contaminant higher than the most conservative RGO. When comparing the least conservative LOAEL-based wildlife RGOs with site concentrations, samples with concentrations higher than the RGOs are clustered in the area just north and south of University Drive (i.e., locations WC-106, WC-107, WC-108, WC-175, WC-177, and WC-216) with the exception of WC-109 and WC-111. When comparing RGOs calculated using mean site-specific BCFs, the results are the same, with the exception of WC-111 having concentrations below the RGO.

8.4.3.3 Sediment Mediated Exposures

Endpoints that addressed the sediment mediated exposures were Assessment Endpoints:

- No. 4 – Protection of Piscivorous Mammals;
- No. 5 – Protection of Piscivorous Birds; and
- No. 6 – Protection of Benthic Macroinvertebrate Communities.

For Assessment Endpoints Nos. 4 and 5, slight risk is demonstrated for lead (e.g., maximum case LOAEL-based HQ = 1.1; Tables 7-15 and 7-17). Food web-based RGOs were not calculated for sediment-mediated exposures.

However, given the magnitude of the HQs based on the piscivore models integrated with the conservatism in the risk estimate (e.g., actual lack of fish, assumption of an FI of 1 from the wetlands/creek), it is unlikely that piscivorous wildlife population is experiencing adverse effects from COPCs in sediment.

For Assessment Endpoint No.6, based on the comparison of sediment concentrations to ATVs and the results of the solid-phase acute toxicity tests with *H. azteca*, adverse effects to the benthic community may be occurring. When comparing site-specific sediment concentrations to ATVs, 80 locations had HQs for arsenic exceeding unity, with 26 locations having lead and/or pesticide concentrations exceeding unity (see Table 8-23).

The toxicity tests found significant adverse effects on survival and growth from exposure to the site sediments from certain locations (i.e., WC-105, WC-160, and WC-175 for survival and WC-106 and WC-109 for growth). These results are inconsistent with the benchmark comparisons as noted below.

Location	Toxicity Observed in <i>H. azteca</i>	Toxicity Predicted by ATV Comparisons		
		Arsenic	Lead	Pesticides
WC-105	Yes (survival)	No	No	Yes
WC-106	Yes (growth)	Yes	Yes	Yes
WC-109	Yes (growth)	Yes	No	No
WC-115	No	No	No	No
WC-118	No	No	No	No
WC-160	Yes (survival)	Yes	Yes	Yes
WC-174	No	Yes	No	Yes
WC-175	Yes (survival)	Yes	Yes	Yes
WC-177	No	Yes	Yes	Yes

Because site-specific toxicity tests are more representative of actual conditions, it can be inferred that the site sediments are less toxic than the ATV comparisons would imply. The sediment ATVs used in this risk assessment and the site-specific toxicity test NOECs and LOECs are presented below.

COPC	Sediment Concentration (mg/kg)	
	ATV*	NOEC-LOEC
Arsenic	57	200-330
Lead	130	260-380
DDT and metabolites	0.570	1.03-2.07
Toxaphene	0.032	6.7-8.7

*ATV for arsenic from Washington State (2008) and for rest, FDEP (2003).

Employing the NOEC-based RGO of 200 mg arsenic/kg would indicate cleanup in the vicinity of approximately 24 locations, all of which are clustered in the area just north and south of Ira Hicks Boulevard, with the exception of two samples much further downstream (WC-155 and WC-196).

Neither lead nor DDT_r (two other COPCs for which *H. azteca*-based NOECs were developed) had concentrations exceeding their respective NOECs at locations where arsenic was less than 200 mg/kg. Toxaphene had only one sample (location WC-177) with concentrations exceeding its NOEC where arsenic was less than 200 mg/kg.

If the sediment toxicity LOEC-based RGO of 330 mg arsenic/kg were employed, cleanup in the vicinity of approximately eight locations, all of which are clustered in the area just north and south of Ira Hicks Boulevard with the exception of one sample much further downstream (WC-155). Lead, DDT_r, and toxaphene (i.e., the three other COPCs for which *H. azteca*-based LOECs were developed) had

concentrations exceeding their respective LOECs at locations where arsenic was less than 330 mg/kg (one location, seven, locations, and 3 locations, respectively).

These facts support the potential use of the 200 mg/kg arsenic RGO as an applicable cleanup level for all OU5 sediment.

8.4.3.4 Surface Water Mediated Exposures

Endpoints that addressed the surface water mediated exposures were Assessment Endpoints:

No. 7 – Protection of Fish Populations; and

No. 8 – Protection of Reptiles and Amphibians.

For Assessment Endpoint No. 7, three lines of evidence were used, two of which (i.e., comparisons to ATVs and *C. dubia* toxicity test results) indicate that there may be unacceptable risks to the fish community at the site (the comparison of residue concentrations with fish TRVs was the exception). However, the results of the media concentration comparisons and the toxicity tests are confounded by the lack of site-specific dissolved metals concentrations and hardness data, the small number of pesticide detects coupled with detection limits higher than ATVs, and pH values lower than the NRWQC recommended values.

Assessment Endpoint No. 8 was assumed to be accounted for by the results considered for Assessment Endpoint No. 7.

Only three non-reference surface water locations were able to be sampled (i.e., WC-105, WC-115, and WC-118). Of these, only two locations had fish present. Sample WC-105 is located at the south end of Spillers Road – no fish were present at the time of the field effort. The other two samples, where fish were present/collected are 2 to 3 miles south of Ira Hicks Boulevard. Although there were issues with detection limits, WC-105 was the only sample in which pesticides were detected. Arsenic and lead were detected in WC-115, but at concentrations below or close to the ATV (HQs = 0.04 and 1.5, respectively). Soil concentrations in the vicinity of WC-115 (i.e., locations WC-116 and WC-117) were not among those contributing to unacceptable soil-mediated risks. In addition, sediment concentrations from locations WC-115 and WC-118 did not elicit toxic effects in *H. azteca*.

Overall, some adverse effects to the aquatic community may be possible at the more upstream locations; but due to the small number of water/fish samples able to be collected and detection limit issues, the extent and causation are not readily apparent. Lack of habitat (i.e., the drying up of the wetlands and creek bed) and high pH in the existing waters may be greater issues in the maintenance of a healthy aquatic community than the COPC concentrations.

8.5 UNCERTAINTY ANALYSIS

There are several sources of uncertainties associated with an ERA, from sampling through chemical analyses, screening assessment, and estimates of toxicity to ecological receptors. Every step in the ERA process involves assumptions which continue through to the final estimates of risk. This is because of the complex interactions of the environmental conditions with the different matrices, different chemicals, and their concentrations, and the different receptors in the environment. The following sections discuss some of the primary uncertainties associated with the risk estimates and the general ERA process.

Uncertainties Associated with COPC Refinement

- Benchmarks for toxaphene parlars were not available; therefore, the potential impacts to the ecological receptors at the site could not be evaluated for those chemicals.
- Additional chemical classes (i.e., volatiles, semivolatiles, and PCBs) were analyzed in the most recent soil/sediment samples. Comparisons of maximum concentrations were presented in Tables 8-1 through 8-4. For soil, only concentrations of 3,3'-dichlorobenzidine and total high molecular weight PAHs exceeded their respective benchmark. However, the 3,3'-dichlorobenzidine was detected in only 1 of 3 samples and the HQ based on the detected concentration was only 1.1. For total high molecular weight PAHs, only one concentration out of three exceeded the soil benchmark with an HQ of 1.6. Given the number of detections, the magnitude of the HQs, and that the benchmarks are conservative screening values, it is unlikely that population-level effects from ecological receptor exposure to these two chemicals in soil-mediated pathways is occurring.

For sediments, only 3,3'-dichlorobenzidine, acetone, and total PAHs exceeded their respective benchmark. However, 3,3'-dichlorobenzidine and acetone were detected in only 1 of 6 samples, generating HQs of 5.6 and 5.8, respectively. PAHs were detected in 8 of 12 samples, with only 1 of the 8 having a total PAH concentration exceeding the benchmark (HQ = 2.2). Given the number of detects, the magnitude of the HQs, and that the benchmarks are conservative screening values, it is unlikely that population-level effects from ecological receptor exposure to these three chemicals in sediment-mediated pathways is occurring.

Uncertainties Associated with Selection of TRVs

- The selection of TRVs is conservative in nature and the values may not actually reflect real-life effects. For example, some of the TRVs were obtained under controlled laboratory conditions, which do not reflect weathered environmental conditions or wild animal behavior patterns. Also, some of the TRVs were extrapolations of toxicity results (e.g., NOAELs to LOAELs, LC50 to NOAELs, etc.) where the selected endpoints were not available. These extrapolations may confound the results and lead to

further uncertainty.

- Most of the literature-derived toxicity values were reported as the concentration in food. If the body weights were not provided, they were assumed or obtained from the literature when the concentrations were converted to doses (i.e., mg/kg BW/day). This practice introduces uncertainties in the derivation of NOAEL and LOAEL TRVs.
- There are also uncertainties associated with the use of other effects data such as the NRWQC. These criteria are supposed to be protective of 95 percent of the sensitive species most of the time; therefore, the remaining 5 percent are not protected.
- Physical and chemical forms of metals like arsenic affect exposure, bioavailability, and subsequent toxicity and are influenced by physicochemical environmental conditions (e.g., pH, TOC, clay content). The specific forms of arsenic and lead at the site were not determined and were assumed to be the same as those used to develop the ATVs and TRVs.
- The avian TRVs for toxaphene are based on a study that used technical toxaphene (TT), which is a non-degraded form of the compound. The mammalian TRVs are based on a study that used weathered toxaphene (WT) which has gone through some of the natural degradation processes. Since the toxaphene present in site soils, sediments, and tissues has been present in the environment since the 1980's, it is likely in a weathered state and therefore WT is a more appropriate form for use in the risk assessment. In general, TT is considered more toxic than WT due to the presence of numerous highly chlorinated congeners that frequently are lost through natural degradation processes. However, the mammalian toxicity study used to develop the TRVs (Besselink et al. 2008) used WT that was extracted from cod liver oil after a two-month exposure and some experts believe that some congeners were present in the cod liver oil at higher concentrations than would be expected if allowed to degrade for a longer (more representative) period. Therefore, it is assumed that the mammalian TRV is very conservative.

Uncertainties Associated with Exposure Estimates

- Several assumptions were made in the food-web models used in the risk characterization (e.g., bioavailability was assumed to be 100 percent for all chemicals). These models have their own inherent uncertainties which contribute to the overall uncertainties in the risk characterization.
- The use of the mean and maximum contaminant concentrations in the food-web models and other lines of evidence to estimate risk may overestimate risk because the individual receptors may not be exposed to these concentrations.

- The risk characterization assumes that receptors are exposed to one chemical at a time and the additive, synergistic, or antagonistic effects of chemicals are not taken into consideration. This assumption may result in the over- or underestimation of risks at the site.
- Uncertainty exists when detection limits are greater than ATVs. Although the chemical may not be detected, the detection limit may be elevated due to interferences or the analytical method may not achieve low enough detection limits. This increases the uncertainty in the risk assessment with these types of detection limits.

Uncertainties Associated with Selection of Assessment Endpoints

- Different species respond differently to chemical challenge. Because it is impractical to use all of the individual species at the site to evaluate risks, the selection of assessment endpoints focuses the risk assessment on the ecosystem components that are most likely to be affected by the site contaminants. This may result in risk estimates that are not protective of certain receptors.

Uncertainties Associated with Establishment of Measurement Endpoints

- Different species respond differently to chemical challenge. Because it is impractical to use all of the individual species at the site to evaluate risks, the selection of assessment endpoints focuses the risk assessment on the ecosystem components that are most likely to be affected by the site contaminants. The representative species selected may not be the most sensitive and TRVs may not be derived from the most sensitive species.

Uncertainties Associated with Site-Specific Toxicity Tests

- Site-specific toxicity tests were used in the risk estimates; however, toxicity test results were not available for all of the potential receptors at the site.
- The samples used in the toxicity tests contained a mixture of chemicals including unknown or tentatively identified compounds which may have contributed to some of the observed effects. The synergistic, antagonistic, additive, and other effects of the COPCs were not determined. Therefore, assuming that sample toxicity is due to a single chemical alone may under- or overestimate the risks at the site.
- The soil toxicity test endpoints were not long enough to determine the long-term effects of the COPCs on the reproduction of *E. foetida*.
- Extrapolations from single-species laboratory bioassays to environmental population-level effects are highly uncertain due to the numerous environmental conditions and interactions that cannot be

replicated in the laboratory (e.g., interspecies interaction like predation).

- Laboratory tests are limited in the number of effects that can be measured; therefore, many biological/physiological effects that could have population-level effects are not assessed.
- Laboratory tests using sediments with multiple COPC often yield results do not follow a strict concentration gradient for any specific chemical (i.e., while *H. azteca* mortality in the 2010 test was highest in sample WC-160, the highest arsenic concentration was observed in sample WC-175, which had slightly lower mortality); therefore, interpretation of cause and effect was equivocal.
- Statistical hypothesis testing used to establish NOECs and LOECs does not result in attribution of causality and significant statistical results do not always equal ecological significance.

Uncertainties Associated with Estimation of Risk

- Some of the calculated HQs were greater than unity when based on the NOAEL but not the LOAEL. Because the NOAEL is based on a concentration that is known to not cause any adverse effects, it makes the interpretation of the results difficult.
- Some chemicals from the background locations showed similar unacceptable risks as those from the site when compared to benchmark values. This makes data interpretation difficult.
- Plants at the site were not collected and chemically analyzed; therefore, plant uptake of COPCs at the site is unknown.
- Total metals surface water concentrations were compared with dissolved metals water criteria. This would result in an overestimate of risk. However, what would be a clear overestimation is confounded by the fact that the lead criterion is based on hardness, and hardness values were not available for the surface water samples. Instead a default hardness of 100 mg CaCO₃/L was assumed, which may or may not reflect conditions at the site.
- Soil ATVs were based on soil concentrations, but not necessarily earthworm concentrations (e.g., SSL for lead was based on avian, not on soil invertebrates). The effect on the risk estimate could be either an over- or an underestimation for the soil invertebrate receptor.
- It was assumed that aquatic community benchmarks would be reasonable surrogate endpoints for evaluating effects to reptiles and amphibians. To the extent that there are differences in toxicity to COPCs between the aquatic community and herptiles, risks to the latter may be under- or overestimated.

8.6 SUMMARY AND RECOMMENDATIONS

The results of the risk characterization indicate that there may be unacceptable risks to ecological receptors from exposure to contaminated soils/sediment in OU5, but not necessarily from surface water or aquatic biota as summarized below.

Assessment Endpoint	Line of Evidence	Potentially Significant COPC-related Ecological Risk?
1 – Protection of Soil Invertebrates	Soil concentrations compared with ATVs	Yes
	<i>E. foetida</i> toxicity tests	No
2 – Protection of Insectivorous, Omnivorous, and Carnivorous Mammals	Food-web based HQs	Yes
3 – Protection of Insectivorous, Omnivorous and Carnivorous Birds	Food-web based HQs	Yes
4 – Protection of Piscivorous Mammals	Food-web based HQs	No
5 – Protection of Piscivorous Birds	Food-web based HQs	No
6 – Protection of Benthic Macroinvertebrate Communities	Sediment concentrations compared with ATVs	Yes
	<i>H. azteca</i> toxicity tests	Yes
7 – Protection of Fish Populations	Surface water concentrations compared with ATVs	No
	<i>C. dubia</i> toxicity tests	No
	Fish residue concentrations compared with ATVs	No
8 – Protection of Reptiles and Amphibians	Accounted for in Endpoint #7.	

Removal of areas of localized contamination may eliminate the major sources of contamination from the sediments and reduce future downstream contamination from erosion/sediment transport to aquatic habitats.

The 200 mg arsenic/kg RGO is recommended as the most appropriate cleanup value to consider as it is a site-specific bounded NOEC value, as opposed to the other RGOs which are based on endpoints that consider only site-specific bioaccumulation and not site-specific toxicity.

Employing the sediment toxicity test NOEC-based RGO of 200 mg arsenic/kg would indicate cleanup in the vicinity of approximately 24 locations, all of which are clustered in the area just north and south of Ira Hicks Boulevard, with the exception of two samples much further downstream (WC-155 and WC-196). Examination of lead and organic pesticide concentrations that would remain in site sediments if the cleanup were to be based on the 200 mg arsenic/kg value indicate that the use of the 200 mg/kg arsenic RGO as an applicable cleanup level for all OU5 sediment would likely be supported.

Assuming that the sediment RGO of 200 mg arsenic/kg is a potential cleanup value, it is important to determine if this value also would protect terrestrial receptors exposed to soil remaining at the site. Assuming the “soil” locations with >200 mg arsenic/kg were removed, average soil concentrations for lead would be higher than the terrestrial food-web-based RGO range; whereas, the remaining COPCs would be within the range of or below the minimum RGO. Although the 200 mg arsenic/kg RGO is a sediment-based value, the areas of concern are drainage areas that are seasonally dry, if at all. As such, the availability of terrestrial or “soil” habitats is not consistent, making the food-web-based soil RGOs overly conservative in the protection of terrestrial-based individuals and populations.

The burden of selecting the final cleanup goals for any site rests with the site risk manager who must balance risk reductions associated with cleanup of contaminants with potential impacts of the remedial actions themselves. For this risk characterization, all of the available information was used to calculate NOEC- or NOAEL- and LOEC- or LOAEL-based RGOs for the protection of invertebrates, mammalian, and avian receptors exposed to COPCs via surficial soils and sediment at the site.

9.0 PHYTOREMEDIATION TREATABILITY STUDY

Phase I – Bench-scale Greenhouse tests were conducted for the Woolfolk site. The bench-scale tests were the initial phase of a two-phase Phytoremediation Treatability Study to evaluate the removal of arsenic and/or pesticides from sediments at the Woolfolk site. The Phase I greenhouse tests were designed to provide data necessary to evaluate the feasibility of proceeding to Phase II of the study: Field Pilot-Scale Phytoremediation Tests. The bench-scale tests were performed at greenhouse facilities located at Fort Valley State University (FVSU) and supported by FVSU staff and students.

9.1 PROJECT MANAGEMENT

9.1.1 Project Objectives and RGOs

The objective of the tests was to evaluate the feasibility of using phytoremediation to treat arsenic- and/or pesticide-contaminated sediments at the site. An acceptable phytoremediation remedy should be capable of achieving the remedial goals of the site where the ecological RGOs for arsenic in surface sediments range from 7 mg/kg for the most sensitive ecological receptors to 10,200 mg/kg for the least sensitive receptor. The proposed RGO for arsenic in site sediments is 200 mg/kg (J. M. Waller, 2010).

9.1.2 Background

Phytoremediation is broadly defined as the use of plants to clean up contamination. Six primary remediation mechanisms have been identified for phytoremediation:

- Phytoextraction
- Phytodegradation
- Phytovolatilization
- Rhizodegradation
- Phytosequestration
- Phytohydraulics

Phytoextraction is the primary mechanism applicable to this treatability study. This mechanism, which is typically used to address inorganic contaminants such as metals, metalloids, and radionuclides, involves contaminant uptake by plant roots, with subsequent accumulation in plant tissue. The plants may require harvesting and disposal to avoid re-contaminating the soil when the plants die or senesce. Several plant species have been identified that will extract high concentrations of metal contaminants (i.e., hyperaccumulate) into their root, stem, and/or leaf tissue. Although ferns have not been shown to hyperaccumulate pesticides, the ferns grown in the Woolfolk sediments during this study were analyzed for pesticides after 12 weeks to determine if any significant uptake occurred.

9.1.3 Previous Phytoremediation Investigations

Integrated Laboratory Systems, Inc., (ILS) conducted a Pilot Study to investigate the bioaccumulation of arsenic in ferns grown in soils/sediment from the Woolfolk site (ILS, 2010). ILS initially tested native plant species growing at the site to determine if they were hyperaccumulating arsenic. Greenhouse tests were also conducted to evaluate the effectiveness of arsenic phytoextraction by Cretan Brake fern (*Pteris cretica* Mayi), Ladder Brake fern (*Pteris vittata*), and Ebony spleenworts (*Asplenium platyneuron*), a plant species native to the site. Of the three plants tested, the Cretan Brake fern was most effective at reducing arsenic concentrations in soils. Although the Ladder Brake fern has been shown in many studies to be a very effective arsenic hyperaccumulator, the plants were not effective in this test. ILS observed that the plants did not grow well during the tests, and concluded that the poor performance was due to transplanting shock.

9.1.4 Arsenic in the Environment

Arsenic has four valence states (0, -3 [arsenide], +3 [arsenite] and +5 [arsenate]) which are generally found in the environment as either arsenite (As^{+3}) or arsenate (As^{+5}). Arsenite is more toxic to humans than arsenate and is generally found under reducing conditions (e.g., flooded soils). The less toxic arsenate is associated with oxidizing conditions, but both forms are often found together as well. Both of these forms of arsenic will adsorb to common soils/sedimentary material such as iron oxides, aluminum oxides, clay minerals, and other soils/sedimentary material such as calcium carbonates and organic matter (Gonzaga et al., 2006). Arsenate adsorbs strongly to iron-oxide surfaces in acidic and near-neutral-pH water (Dzombak and Morel, 1990; Waychunas et al., 1993); however, as the pH increases to 8 or above, arsenate will desorb (Fuller and Davis, 1989; Dzombak and Morel, 1990). Weaker sorption occurs between arsenate and aluminum oxide and clay minerals, and arsenite weakly sorbs to all three of these materials, but pH-dependent adsorption and desorption reactions will likely control the mobility of arsenite and arsenate in the environment (Manning and Goldberg, 1997). Other ions in soils/sediments or groundwater can control the mobility of arsenic. Phosphate is an analog of arsenate and will compete with it for sorption sites in sediments (Manning and Goldberg, 1996). Oxyanions such as molybdenum, selenium, and vanadium may also compete with arsenic for sorption sites as well (Robertson, 1989).

9.1.5 Phytoremediation of Arsenic

Several ferns of the genus *Pteris* have been found to hyperaccumulate arsenic. These include Chinese Brake fern or Ladder Brake fern (*Pteris vittata* L.), the thin leafed Brake fern or parlor fern (*Pteris biaurita* L.), the striped Brake fern (*Pteris quadriaurita* Retz.), and *Pteris ryukyuensis* Tagawa (Ma et al., 2001). The ferns were discovered to hyperaccumulate arsenic into their fronds, stalks, and roots (Chen and Wei, 2000; Ma et al., 2001). Other arsenic hyperaccumulating plants include Dixie Silverback fern (*Pityrogramma calomelanos*), Indian mustard (*Brassica juncea* and *Brassica nigra*), velvet grass (*Holcus*

lanatus), hemp sesbania (*Sesbania exaltata*) pitted morning glory or white morning glory (*Ipomoea lacunosa*), and Chinese cabbage (*Brassica pekinensis* Rupr.). Brake fern was proposed for this site because it is very tolerant to high concentrations of arsenic (500 mg/kg and higher), and it accumulates most of the arsenic (75 to 99 percent) in its above ground biomass which makes it easier to harvest. Silver Ribbon ferns (*Pteris cretica*) were proposed because they have shown the ability to hyperaccumulate arsenic (Kissell, et al., 2003) and would acclimate to the Woolfolk climate (Michael Blaylock, President of Edenspace Systems, Inc., personal communication, 2011).

As discussed above, pH and the presence of various ions will affect the mobility of arsenic in the environment. These factors will influence the bioavailability of the contaminant to the plant which directly affects the effectiveness of phytoremediation. In particular, phosphate, an analogue of arsenate, and heavy metals can influence the effectiveness of arsenic accumulation by ferns. When arsenic concentrations are high (approximately 400 mg/kg) the presence of phosphate may improve phosphate nutrition in the plants and enhance arsenic accumulation, while at lower arsenic concentrations (50 to 200 mg/kg) arsenate can interfere with phosphate metabolism and have toxic effects on the plant (Tu and Ma, 2003a). Tu and Ma (2003a) concluded that a minimum 1:2 molar ratio of phosphate to arsenic in the soil solution would enhance arsenic accumulation. Boisson et al. (1999) reported that increased uptake of arsenic by plants in soils treated with hydroxyapatite may be due to the displacement of adsorbed arsenic by the phosphate added to the soils. Cao, et al. (2003) reported that the addition of phosphate to a site contaminated by chromate-copper-arsenate enhanced plant growth and also mobilized arsenate due to competitive anion exchange. Therefore, it is important to realize that, while the addition of phosphate to the system may increase plant growth; it may also mobilize arsenic which can be transported to the groundwater. The effects of arsenic mobilization can be countered by using a fast growing plant which develops an extensive root system such as Ladder Brake fern (*Pteris vittata* L.) (Gonzaga et al. 2006).

The effects of soil and water pH can also affect arsenic bioavailability. Tu and Ma (2003b) studied arsenic and phosphate uptake by Brake ferns grown hydroponically. Their results indicated that the highest arsenic uptake was observed at a pH less than 5.21. Bagga and Peterson (2001) adjusted the pH of soil containing 300 mg/kg arsenic to 4.5, 5, 6 and 7. The Asparagus fern was most effective at removing arsenic from soils buffered to pH 5.

With the exception of lead, other metals including cadmium, zinc, nickel, and copper inhibited the uptake of arsenic by ferns (Gonzaga et al., 2006). In general arsenate is less mobile in acidic soils with high amounts of clay and oxides while arsenite is the more mobile and toxic moiety.

The following generalizations can be made regarding phytoremediation of arsenic:

- Arsenate will be more abundant in oxidized (dry and aerated) soils while arsenite will be the dominant species in reduced soils (wet and more anaerobic).
- Arsenate will be more mobile at higher pH (>8).
- Arsenite is more mobile than arsenate and its mobility increases at lower pH (<7).
- Metals ions such as cadmium, copper, nickel, and zinc in soils will decrease the effectiveness of phytoremediation of arsenic by the ferns, and other metals in soils such as molybdenum, selenium, and vanadium will compete with arsenic for sorption sites in soils.
- Addition of phosphate to the soils can increase plant growth and arsenate uptake, but can also result in increasing the mobility of arsenic in soils which can be transported to the groundwater.
- The addition of compost to the soils can increase plant growth, but arsenic may be bound to organic material in the compost which will inhibit plant uptake (Cao et al. 2003).

9.1.6 **Phytoextraction Mechanisms**

Several mechanisms have been identified during phytoextraction of arsenic by the Brake fern.

- Uptake of arsenate (As^{+5}) by the plant followed by reduction to arsenite (As^{+3}) in the plant. This transformation of arsenic is important due to the increased toxicity of arsenite relative to arsenate.
- Translocation of arsenic from roots to shoots via xylem sap.
- Free arsenic in the cytoplasm is chelated and bound to the plant cell walls by phytochelatins.
- Vacuolar storage reduces free arsenic in cytoplasm.
- Mycorrhizal symbiosis (between fungus and plant root) enhances nutrient absorption area and uptake kinetics.

9.1.7 **Plant Selection**

The Chinese Brake fern (*Pteris vittata* L.) was used in these tests due to the many case studies supporting its effectiveness at phytoextracting arsenic from sediments. The Silver Ribbon fern (*Pteris cretica* 'Albo-lineata') was chosen because it has also shown the ability to hyperaccumulate arsenic (Kissell, et al, 2003). Another benefit of these plants is that as hardy perennials, they can be reused, and studies have shown that ferns can be harvested as many as three times in a six month period (Natarajan et al., 2008). Both species are non-native to the U.S., but the Chinese Brake fern is distributed throughout the southeastern U.S. as far north as South Carolina, and the Silver Ribbon Brake fern is present throughout Florida, but should survive in Georgia as well.

A previous Pilot study conducted by ILS using soils from the Woolfolk site indicated that native species were not effective at reducing arsenic contamination in the soils at this site. They also stated that the Brake fern *Pteris vittata* was not effective in their study, but attributed this result to transplant shock. To prevent transplanting shock, an older more mature plant stock was used in the tests.

9.1.8 Fate of Contaminants in the Plant Tissue

The Chinese Brake fern and Silver Ribbon ferns were used in these tests due to their ability to hyperaccumulate arsenic and store most of the extracted arsenic in their fronds. The location of the arsenic in the aboveground fronds makes harvesting easier. For Phase II testing this will also allow plants to harvest fronds multiple times during a growing season instead of sacrificing whole plants for analysis.

9.1.9 Considerations

The factors that were considered in planning these tests are as follows.

9.1.9.1 Identification of Contaminant(s)

In addition to arsenic, other metals at the site may compete with arsenic for uptake by the plants. If other contaminants at the site are toxic to the ferns, then phytoextraction may be inhibited. An initial analysis of the soil by the University of Georgia (UGA) Soils laboratory did not indicate the presence of any metals or other contaminants that might interfere with uptake and is discussed further in Section 9.3.

9.1.9.2 Concentration of Contaminant(s)

Sediments were collected in areas with relatively high arsenic concentrations (>150 mg/kg) and relatively low concentrations (<75 mg/kg).

9.1.9.3 Soil/Sediment Type

Oxidized sediments and reduced sediments (based on color, odor, and moisture content) were separately collected to evaluate the effect on phytoremediation.

9.1.9.4 pH

Lime was added to the sediments to adjust the initial pH (pH <5) to a pH of approximately 6, which is recommended for optimal plant growth/health.

9.1.9.5 Average High Temperature

The daily average high temperature was monitored in the controlled greenhouse environment. Measured temperatures ranged from a low of 62° to a high of 90° F.

9.1.9.6 Soil Nutrient Levels Phosphorous, Potassium, Nitrogen and Organic Matter Levels

The nutrient levels were tested and amended using a weak solution of MiracleGro™ fertilizer to support plant growth. Triple superphosphate (TSP) was added to selected treatments to determine if it might increase arsenic phytoextraction.

9.1.9.7 Soil Moisture and Plant Watering

Soil moisture was monitored daily using a direct-reading probe and the plants watered as needed.

9.1.9.8 Daily Illumination

The selected ferns prefer bright indirect sunlight. The greenhouse was covered with a 70 percent shade cloth to provide indirect light, and a light meter was used to measure light intensity in the greenhouse each day.

9.1.9.9 Average Relative Humidity

The selected ferns prefer a warm humid climate. A lack of humidity was not a factor at the Woolfolk site, but leaves were misted daily during the greenhouse tests.

9.2 PHASE I GREENHOUSE STUDY

9.2.1 Study Objective

The study objective was to evaluate the effectiveness of phytoremediation of arsenic and/or pesticide-contaminated sediments at the Woolfolk site using Chinese Brake fern and Silver Ribbon fern under greenhouse conditions.

The specific goals of the greenhouse tests were to:

1. Demonstrate the viability of using arsenic hyperaccumulating ferns to extract arsenic from soils/sediments at the Woolfolk site.
2. Compare the relative effectiveness of the Chinese Brake fern (*Pteris vittata* L.) and the Silver Ribbon fern (*Pteris cretica* 'Albo-lineata') at lowering arsenic concentration in sediments.

3. Evaluate the translocation of arsenic within the plant tissue.
4. Evaluate plant health during the greenhouse study to determine if site sediments are capable of supporting fern growth.
5. Compare the use of mature ferns to the results of a previous study by ILS that used young plants (ILS, 2010).

9.2.2 Task 1 – Bench-Scale Laboratory/Greenhouse Optimization Study

9.2.2.1 Sample Collection and Characterization

The bulk soil/sediment samples were collected on March 21 and 22, 2011 by the J. M. Waller team. Sediments were screened in the field using a portable hand-held XRF meter to estimate arsenic concentration. Four 5-gallon buckets (a minimum of 25 kg [100 kg total]) of soil were collected for each of the following soil types and labeled as follows:

- WC-300 - Well drained oxidized sediments with low arsenic concentration (< 75 parts per million [ppm] arsenic)
- WC-244 - Well drained oxidized sediments with high arsenic concentration (> 150 ppm arsenic)
- WC-204 - Poorly drained reduced sediments with low arsenic concentration (< 75 ppm arsenic)
- WC-216 - Poorly drained reduced sediments with high arsenic concentration (> 150 ppm arsenic)

Drier, lighter colored (light brown, brown, yellowish brown, and red), well aerated sediments were characterized as oxidized, while darker (gray, black and bluish), wetter sediments were characterized as reduced. Samples were collected in 5-gallon plastic buckets, sealed, labeled, and stored in a greenhouse until use.

9.2.2.2 Laboratory/Greenhouse Design and Setup

The treatment groups are listed in Table 9-1. These include eight replicates that were set up for each treatment and controls were set up in quadruplicate. Half of the treatments (four pots per event) and control pots (two pots per event) were sacrificed at six weeks and the remaining half were sacrificed at 12 weeks. Samples of the fern fronds, roots, and sediments were separately composited and submitted to the USEPA Region 4 SESD Laboratory in Athens, Georgia for analysis. Composite soil samples were collected from the control pots at 6 and 12 weeks as well.

Sediments were placed in an FVSU greenhouse and spread on plastic sheeting to dry. After drying, the sediments were manually homogenized and composite samples were collected on April 1, 2011. Sampling consisted of collecting 20 random grab samples from each soil type and placing them in a sealed plastic bag. The contents of the bag (approximately 2 kg) were then manually homogenized. The bags were labeled with the sample identification. One split of each sample was submitted to a laboratory for a soils analysis (Table 9-2), and a second split of each was submitted to a laboratory to determine the total arsenic concentration using USEPA Method 3050 for sample digestion with analysis by Method 6010B Inductively Coupled Plasma-Atomic Emission Spectrometry (Table 9-3). Once the initial analyses were received, the bench scale tests were set up in a greenhouse.

The soil analysis results indicated that sediments were clay and sandy loams. The pH of the sediments varied from 4.10 to 4.55. Although the ferns are often more effective at arsenic removal at a pH of 5, lime was added to the sediments to raise the pH to approximately 6 based on the UGA Soils laboratory recommendations for ferns. This was done to assist plant health which was of concern due to the plant mortality reported in the previous tests by ILS, 2010. Phosphorus levels varied from 3.94 to 41.03 mg/kg which was 1 to 2 orders of magnitude below the targeted phosphorus treatment concentrations. Additional phosphorus was added to increase the phosphorus concentrations to approximately 200 mg/kg. All other laboratory results indicated that these sediments were sufficient to support fern growth without additional amendment; however, the pH was raised to enhance fern growth.

Chinese Brake ferns and Silver Ribbon ferns of the same age and approximate size were acquired via a subcontractor. Larger more mature ferns were used to inhibit transplant shock. J. M. Waller Team personnel returned to the site on April 6, 2011, when the ferns arrived, to further process the sediments and pot the ferns. The soil analysis results included the estimated lime requirement (based on soil volume) needed to raise the soil pH to approximately 6. The sediments were mechanically homogenized using a motorized Multiquip MC94SH8 9-cubic foot concrete mixer for 15 to 30 minutes. The processed sediments were returned to the plastic sheeting and spread out prior to placing in 1.9 liter (L) (½ gallon) pots. Sets of “no fern” controls (four replicates) or treatments (eight replicates) were prepared as listed in Table 9-1. Each replicate consisted of 1.4 L of soil (approximately 2 kg) in a single 2 L plastic pot with a hole at the base.

The soil analysis results included phosphorus concentrations for each soil type. These data were used to calculate the mass of TSP needed to raise the phosphorus concentration of each soil type to 200 mg/kg. The appropriate mass of TSP was added to and manually homogenized into each of the remaining sediments spread out on the plastic sheeting. These sediments were then used to fill an additional 20 pots with 1.4 L of the dried processed sediments (2 kg). The sediments in the pots were then hydrated with 10 ounces of dilute (1/2 strength) MiracleGro™ (20-20-20) fertilizer and water. A plastic water pan was placed under each pot to collect leachate.

Following rehydration, the ferns were potted in the contaminated sediments. The ferns were in 3-inch pots and were root bound upon arrival. Therefore, it was not possible to remove significant amounts of potting soil from the root mass. A core of soil the approximate size of the fern root mass was removed from each pot using a bulb planter. The fern root mass was placed in the cored hole and the cored soil was used to fill in the gaps around the root mass. The ferns were planted so that the top of the root mass was level with the top of the sediments in each pot. The ferns were watered with 10 ounces of tap water to ensure that all of the sediments were hydrated. A deionized (DI) water system was set up at the site on April 12, 2011. Scotts MiracleGro™ fertilizer was added to DI water at a 0.025 percent concentration. This solution was used to mist and water the ferns for the remainder of the study.

On April 17, 2011, a 50 percent shade cloth was installed on the exterior of the greenhouse to shade the ferns. A 72 percent shade cloth was installed approximately one month later.

9.2.3 Laboratory/Greenhouse Study Monitoring

Once the laboratory/greenhouse study was set up, the plants were monitored on a daily basis for 13 weeks (Appendix V – Appendix A). The ferns were watered and misted daily or as needed with either a dilute DI water MiracleGro™ mixture (0.025 percent) or deionized water by a qualified scientist or technician.

Additional fertilizer was not required, but if pests infested the plants, a nonhazardous pesticide was used as a control measure. A list of monitoring parameters is included as Table 9-4.

Plants were watered, but not to exceed field capacity of the sediments. Soil water was not monitored in these tests. Plant tissues were segregated into roots (below ground) and stems/fronds (above ground).

Replicates of plants were selected at random and sacrificed at 6 and 13 weeks prior to laboratory analysis. The roots were gently removed from each pot, and as much soil as possible was returned to the pot. Residual soils were rinsed from the root mass using deionized water. The fronds and stems were cut from the root mass. The root mass from the selected replicates were placed in sealable plastic bag and labeled while the fronds and stems from the selected replicates were placed in a separate sealable plastic bag and labeled. Immediately after collection, all of the samples were placed in ice-cooled insulated chests for shipment to the USEPA laboratory.

The sediments were removed from each of the four treatment pots, composited, and manually homogenized. Discrete samples were collected from the homogenized, composited sediments and submitted to a laboratory for analysis of total arsenic using USEPA Method 6010B and reported in mg/kg (dry wt.).

9.2.3.1 Laboratory/Greenhouse Study Data Analysis and Performance Evaluation

Arsenic removal was calculated by comparing the arsenic concentration in the treated samples with the control samples. Plant biomass was visually monitored to evaluate the effects of arsenic on plant growth and to determine watering and fertilization needs.

9.3 RESULTS

The greenhouse bench-scale test results are presented below.

9.3.1 Comparison of the Use of Mature Ferns to the Results of a Previous Study by ILS Using Woolfolk Sediments

Plant growth and arsenic uptake were hampered in the previous study conducted by ILS (2010). This result was attributed to transplant shock which was the result of using less mature plants. The use of more mature plants in this study reduced the potential of this happening. No plant mortality was observed during the duration of the experiment and all plants appeared healthy prior to sacrifice.

9.3.2 Effects of pH and Phosphorus Concentration in Sediments on Arsenic Removal

After 13 weeks, pH concentrations in the sediments ranged from lows of approximately 5.7 to highs of 7.2 (Table 9-5). This relatively narrow range of pH was not considered statistically significant and no patterns were observed which correlated higher or lower pH to arsenic removal. The pH levels measured in control sediments were very similar to those measured in the treated sediments.

Phosphorus concentrations in the same sediments varied from 14.4 mg/kg to 38.2 mg/kg in untreated sediments and from 85.5 mg/kg to 274.2 mg/kg in sediments treated with TSP. At the lowest initial arsenic concentrations in sediments, (13 to 19 mg/kg in the WC-300 samples), a higher concentration of arsenic was removed in the sediments treated with phosphorus. In two treatments, Silver Ribbon fern treatments WC-204 and WC-244, the percentage of arsenic removed decreased. However, in all of the other treatments the addition of TSP correlated with a lower percentage of arsenic removal relative to sediments that did not receive additional phosphate (Table 9-6).

9.3.3 Plant Health During the Greenhouse Study to Determine if Site Sediments Have the Capability of Supporting Fern Growth

Plants were monitored daily and photographed weekly to chart growth patterns. As the study progressed for 13 weeks, some lesions developed on fronds and some fronds dried out and died, but no plants died during these tests. The dead leaves were attributed to excessive exposure to sunlight which was more intense in some sections of the greenhouse at certain times of the day. During harvest, the rhizomes appeared healthy and all appeared to have increased in mass as the tests continued.

It was not possible to quantify the root mass due to the difficulty in removing soil from the rhizome. However, photographs were taken weekly during the course of the project and provide qualitative data on the increase in plant biomass (Appendix V – Appendix B). Photos of the ferns include an initial set (Figure B1, Appendix V – Appendix B) and photos of Silver Ribbon ferns and Chinese Brake ferns grown in sediments with low concentrations of arsenic (Figures B2 through B7, Appendix V – Appendix B) and the same fern species grown in sediments with higher arsenic concentrations (Figures B8 through B13, Appendix V – Appendix B). The photos of the ferns when initially planted show plants with approximately six fronds (Figures B2, B4, B8, and B11, Appendix V – Appendix B). After 13 weeks the fronds on both species have approximately doubled in number and volume (Figures B5, B7, B10, and B13, Appendix V – Appendix B).

Photographs of the root mass also show growth. Roots were visible at the soil/pot interface in all plants as they were harvested at 13 weeks (Figure B14, Appendix V – Appendix B). The root ball in all plants had also increased in mass and volume during this period. Figures B15 and B16 (Appendix V – Appendix B) show the root mass compared to its approximate initial size after 13 weeks.

9.3.4 Translocation of Arsenic Within the Plant Tissue

Arsenic was primarily translocated to the fronds of the ferns rather than the root mass (Table 9-7). This result is promising for application of ferns at this site because it opens the possibility that the fronds can be harvested multiple times per year rather than requiring replanting.

9.3.5 Arsenic Removal Results

Arsenic concentrations increased in all of the plant fronds as the tests proceeded (Appendix V – Appendix C). Arsenic frond concentrations ranged from a low of 56 mg/kg to a high concentration of 630 mg/kg after 6 weeks growth in the contaminated sediments. After 13 weeks the arsenic frond concentrations ranged from 96 mg/kg to 980 mg/kg. Higher arsenic concentrations were generally observed in ferns planted in the sediments with higher concentrations of arsenic.

The percentage change in arsenic concentrations in the sediments as compared to control concentrations were calculated for the four different treatments of each of the four different sediments (16 individual comparisons) (Tables 9-7 and 9-8). After 13 weeks, arsenic concentrations decreased by as much as 23.8 percent in 9 of the 16 comparisons. Arsenic concentrations were unchanged in 4 of the 16 comparisons, and arsenic increased (4.0 percent to 12 percent) in 3 of the comparisons. The arsenic concentration was initially measured in the sediments prior to addition of TSP. These results are included in the control samples listed in Table 9-3 as WC-204-C, WC-216-C, WC-244-C, and WC-300-C. When these data were compared to the final arsenic concentrations measured in the controls analyses, arsenic

concentrations increased from 4.54 to 30.0 percent indicating that losses in the treatments cannot be attributed to factors other than the presence of the fern plants.

Plans to estimate the efficiency of arsenic removal were cancelled due to the inability to effectively weigh the root mass of the fern plants. Sediments were tightly bound to the dense root mass making it extremely difficult to remove the sediments mass from the rhizome. An accurate root mass could not be obtained; therefore, it was not possible to accurately estimate overall efficiency of arsenic removal by the plant.

9.3.6 Effectiveness of Chinese Brake Fern versus Silver Ribbon Fern

Arsenic concentrations in the Chinese Brake fern fronds were higher on average than the concentrations observed in the Silver Ribbon fern fronds and were as much as 64 percent greater when comparing dry weight concentrations (Table 9-7). However, when comparing percent changes of arsenic in soil (Table 9-8), the Brake fern and Silver Ribbon fern were comparable. They both removed the same percentage of arsenic in two of eight comparisons, the Brake fern outperformed the Silver Ribbon fern in three of eight comparisons and the Silver Ribbon fern outperformed the Brake fern in three of eight comparisons.

9.3.7 Pesticide Uptake by Ferns

Sediments and plant tissues were also analyzed for pesticides. While the sediments had pesticide concentrations in the low parts per billion to low parts per million range, the concentrations of pesticides in the plant tissues were at least an order of magnitude lower. For example, 4,4'-DDT (p,p'-DDT) was detected in WC-244 sediments at 0.6 mg/kg and was detected in the fern fronds at 0.0075 to 0.085 mg/kg. These data indicate that pesticides were not being hyperaccumulated into the plant tissues.

9.4 CONCLUSIONS AND RECOMMENDATIONS

The bench-scale test showed that Chinese Brake ferns and Silver Ribbon ferns removed arsenic from the Woolfolk sediments. Both species of ferns were able to survive and flourish in the sediments under greenhouse conditions. The percentage of arsenic removed from the sediments varied from 0 to 27 percent, but these removal rates would be expected to increase as the plants mature during a normal growing season. There was no clear advantage to amending the sediments with phosphorus. Also there was no apparent advantage to using Chinese Brake fern versus Silver Ribbon fern as they both accumulated arsenic to similar levels. As shown by the bench-scale testing, if arsenic concentrations in the root zone remain above the RGOs this would potentially allow multiple harvesting of fronds while the rhizome is allowed to remain in place.

If phytoremediation is considered a viable remediation method, J. M. Waller recommends conducting a field pilot test to provide the following necessary information prior to full scale implementation:

1. The ability of the ferns to survive under non-greenhouse conditions.
2. An estimation of arsenic removal under non-greenhouse conditions for an entire growing season.
3. The potential for more than one harvest of the ferns per growing season.
4. The ability of the ferns to survive winter conditions in Fort Valley, Georgia.

It is recommended that any pilot testing be started in the early to late spring in order to allow the plants to establish themselves and to provide for multiple harvests of the fronds during the growing season.

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10.0 SUMMARY AND CONCLUSIONS

The OU5 study area is a rural area where sediment may have been impacted by surface runoff from the former Woolfolk facility. This study area, consisting of approximately 400 acres, follows an approximately 3 mile long ephemeral stream within the floodplain of an upper tributary of Big Indian Creek. The study area consists of mostly undeveloped properties (mainly woodlands and wetlands) that are zoned rural agricultural, rural residential, and medium density residential.

This OU5 RI report summarizes the activities performed under various investigation programs by federal agencies and contractors over a 6-year period (2005-2011). The environmental media evaluated for this OU5 study area are sediments. Groundwater, soil, and indoor air have been investigated and reported under other OUs and are not discussed in detail within this RI Report.

The objectives of this OU5 RI Report are to:

- Characterize the nature and extent of contamination in sediments along the ephemeral stream using relevant existing data and collection of additional data as needed;
- Identify and delineate the source of the arsenic detected in the sediments;
- Prepare risk assessments to determine whether metal or pesticide contaminants in sediments pose a current or future potential risk to human health and the environment;
- Identify the potential contaminant migration pathways; and
- Collect sufficient data for development of a limited number of remedial action alternatives that are protective of human health and the environment and that satisfy ARARs.

10.1 NATURE AND EXTENT OF CONTAMINATION SUMMARY

This section summarizes the known nature and extent of sediment contamination, detailed in Section 5.0. The sediment at OU5 contains metals and pesticides. This RI Report addresses the OU5 site area sediment contamination along the ephemeral stream originating at end of Spiller Street extension, generally heading due south for 3 miles, and terminating south of Carver Drive Bridge into an upper tributary of Big Indian Creek. The focus of this RI Report has been on identifying contaminants that are attributable to the former Woolfolk facility.

10.1.1 Nature and Extent of Sediment Contamination

The sediment contamination has been investigated in numerous field events. These investigations have revealed that contaminants of concern are present in the sediment at various concentrations, within the entire OU5 study area. Historically, the Woolfolk facility has been used for the production and packaging of organic and inorganic insecticides, including arsenic and various organic pesticides, including DDT, lindane (BHC), toxaphene, and other chlorinated pesticides

10.1.1.1 Metals

Metals (arsenic and lead) are identified contaminants in the source areas within the OU5 study area. Arsenic and lead exist at concentrations greater than the RGOs in the sediment samples. The metals within OU5 are attributable to the Woolfolk facility. Metal contamination was most noticeable at two locations: the area adjacent to Ira Hicks Boulevard and the area adjacent to Carver Drive. Sediments in these areas had metal concentrations that exceeded the RGOs. The extent of the metal contamination within the study area is presented in Tables 5-1 through 5-3 and on Figures 5-1 through 5-9.

10.1.1.2 Pesticides

Pesticides are identified contaminants in the source areas within the OU5 study area. DDD, DDE, DDT and toxaphene exist at concentrations greater than the RGOs in the sediment samples. The pesticides within OU5 are attributable to the Woolfolk facility. Pesticide contamination was most noticeable at one location: the area adjacent to Ira Hicks Boulevard. Sediments in this area had pesticide concentrations that exceeded the RGOs. The extent of the pesticide contamination within the study area is presented in Table 5-4 and on Figures 5-10 through 5-21.

10.2 CONTAMINANT FATE AND TRANSPORT SUMMARY

Contaminant fate and transport in the environment are controlled by a number of factors including chemical and physical properties of the contaminants, geologic formations, hydrologic conditions, topography, and precipitation.

The sediment contaminants identified in the nature and extent discussion are assumed to be associated with the former Woolfolk facility. The major pathway of migration within the OU5 study area was surface water runoff.

All of the pesticides have low water solubilities, high K_{oc} values, and tend to associate with sediments. It should be noted, however, that association of contaminants with sediments does not preclude the migration of these contaminants since the contaminated sediments may be conveyed downstream from

the drainage ditch to an upper tributary of Big Indian Creek via sediment transport mechanisms during storm events.

As indicated in Table 6-1, the inorganic (metals) contaminants of concern (arsenic and lead), and the organic (pesticides) contaminants of concern (DDD, DDE, DDT and toxaphene) at this site have K_d values greater than 10 mL/g. In general, these contaminants are expected to readily sorb to soil particles at the site. Therefore, the leaching of these contaminants of concern from soils to groundwater at the OU5 site is expected to be relatively limited due to their high propensity for sorption.

10.3 RISK ASSESSMENT SUMMARY

The risk assessment for this RI focused on both human health and ecological risks.

10.3.1 Human Health Risk Assessment

The HHRA identified metals (aluminum, antimony, arsenic, cobalt, iron, lead, manganese, mercury, and vanadium) and pesticides [benzo(k)fluoranthene, chrysene, and indeno(1,2,3-cd)pyrene, DDD, DDE, DDT, alpha-BHC, beta-BHC, delta-BHC, gamma-BHC, endrin aldehyde] as the COPCs for sediment exposures within the study area. These contaminants were selected based on the USEPA Region 4 regional screening level for residential backyards. The table in Section 7.1.4 presents the detected contaminants (metals and pesticides) in each EU that exceeded the residential soil USEPA RSLs. Table 7-10 presents a summary of the metals detected in the background samples. The comparisons of site surface soil metals concentrations to the background values are shown in Table 7-11.

The sediment was the only route of exposure that was evaluated quantitatively and qualitatively in the HHRA. Table 7-24 presents the exposure parameters for trespassers, groundskeepers, and utility workers. Table 7-25 presented the residential exposure parameters. See Tables 7-27 and 7-28 for a summary of the potential risks that could result from exposure to indoor air or future surface water exposures based on sediment concentrations.

With the exception of the residential exposure scenario, the RME and CTE cancer risks for the trespassers, groundskeeper, and utility worker receptors were within or below the USEPA risk range and all of the noncancer HIs were less than the noncancer benchmark of 1.0 (see Tables 7-27 and 7-28 and Figures 7-7 and 7-8). The residential risks exceeded USEPA's applicable cancer and noncancer risk thresholds, but it is unlikely that residential exposure would ever occur in OU5. The residential scenario was evaluated to provide an upper-bound on the potential levels of risk. Given that the overall approach to the HHRA would tend to overestimate actual risks to a fairly significant degree, it is unlikely that exposure to OU5 site soils would result in any unacceptable health impacts for the evaluated non-residential receptors.

10.3.2 Ecological Risk Assessment

The ERA indicates that there may be unacceptable risks to ecological receptors from exposure to contaminated sediment in OU5, but not necessarily from surface water or aquatic biota. Removal of areas of localized contamination may eliminate the major sources of contamination from the sediments and reduce future downstream contamination from erosion/sediment transport to aquatic habitats.

Assuming a sediment RGO for arsenic of 200 mg/kg, remediation would be required in the vicinity of approximately 24 locations. Most of these locations are adjacent to Ira Hicks Boulevard, with the exception of two locations near the Carver Drive Bridge (WC-155 and WC-196).

Assuming the sediment locations with >200 mg /kg of arsenic were removed, average soil concentrations for lead would be higher than the terrestrial food-web-based RGO range; whereas, the remaining COPCs would be within the range of or below the minimum RGO. Although the 200 mg/kg arsenic RGO is a sediment-based value, the areas of concern are drainage areas that are mostly seasonally dry. As such, the availability of terrestrial or "soil" habitats is not consistent, making the food-web-based soil RGOs overly conservative in the protection of terrestrial-based individuals and populations.

10.3.3 Conclusions

The interpretation of the data and information compiled for this RI indicates that:

- Metal and pesticide contamination is present along the ephemeral stream originating at end of Spiller Street extension, generally heading due south for 3 miles, and terminating south of the Carver Drive Bridge into an upper tributary of Big Indian Creek. The contaminants in the sediment at the OU5 site area are attributable to the former Woolfolk facility.
- The HHRA estimates the potential risks associated with the sediment contamination. The residential risks exceeded USEPA's applicable cancer and noncancer risk thresholds, but it is unlikely that residential exposure would ever occur in OU5. The residential scenario was evaluated to provide an upper-bound on the potential levels of risk. Given that the overall approach to the HHRA would tend to overestimate actual risks to a fairly significant degree, it is unlikely that exposure to OU5 site soils would result in any unacceptable health impacts for the evaluated non-residential receptors. With the exception of the residential exposure scenario, the RME and CTE cancer risks for the trespassers, groundskeeper, and utility worker receptors were within or below the USEPA risk range and all of the noncancer HIs were less than the noncancer benchmark of 1.0.

- The ERA indicates that there may be unacceptable risks to ecological receptors from exposure to contaminated sediment in OU5. Removal of areas of localized contamination may eliminate the major sources of contamination from the sediments and reduce future downstream contamination from erosion/sediment transport to aquatic habitats. The 200 mg/kg arsenic remedial goal option is recommended as the most appropriate cleanup value to consider as it is a site-specific bounded NOEC value. Assuming the sediment locations with >200 mg/kg of arsenic were removed; average soil concentrations for lead would be higher than the terrestrial food-web-based RGO range; whereas, the remaining COPCs would be within the range of or below the minimum RGO. As such, the availability of terrestrial or sediment habitats is not consistent, making the food-web-based soil RGOs overly conservative in the protection of terrestrial-based individuals and populations.
- The Phytoremediation Treatability Study showed that Chinese Brake ferns and Silver Ribbon ferns removed arsenic from the Woolfolk sediments under greenhouse conditions. The percentage of arsenic removed from the sediments varied from 0 to 27 percent. As shown by the bench-scale testing, if arsenic concentrations in the root zone remain above the RGOs, this would potentially allow multiple harvesting of fronds while the rhizome is allowed to remain in place. If phytoremediation is considered a viable remediation method, J. M. Waller recommends conducting a field pilot test prior to full scale implementation:

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APPENDIX I

PREVIOUS USEPA INVESTIGATIONS

APPENDIX II

LABORATORY REPORTS

APPENDIX III

HUMAN HEALTH RISK ASSESSMENT REPORT

APPENDIX IV

ECOLOGIC RISK ASSESSMENT REPORT

APPENDIX V

PHYTOREMEDIATION TREATABILITY STUDY REPORT